7. Land-Use Change and Forestry

This chapter provides an assessment of the net greenhouse gas flux[1] resulting from forest lands, croplands, and settlements. IPCC *Good Practice Guidance for Land Use, Land-Use Change and Forestry* (IPCC 2003) recommends reporting fluxes according to changes within and conversions between these land use types, as well as grassland and wetlands. However, consistent datasets are not available for the entire United States to allow results to be partitioned in this way. Therefore, greenhouse gas flux has been estimated for the following categories: 1) forest land remaining forest land 2) croplands remaining croplands, and 3) settlements remaining settlements. This categorization provides additional sources of information regarding N_2O emissions by major land use type.

It should be noted that other land-use and land-use change activities result in fluxes of non-CO_2 greenhouse gases to and from soils that are not comprehensively accounted for currently. These fluxes include emissions of CH_4 from managed forest soils, as well as CH_4 emissions from artificially flooded lands, which result from activities such as dam construction. Aerobic (i.e., non-flooded) soils are a sink for CH_4, so soil drainage can result in soils changing from a CH_4 source to a CH_4 sink, but if the drained soils are used for agriculture, fertilization and tillage disturbance can reduce the ability of soils to oxidize CH_4. The non-CO_2 emissions and sinks from these other land use and land-use change activities were not assessed due to scientific uncertainties about the greenhouse gas fluxes that result from these activities.

The greenhouse gas flux from forest land remaining forest land is reported using estimates of changes in forest carbon stocks and the application of nitrous oxide (N_2O) fertilizers to forest soils. Seven components of forest carbon stocks are analyzed: aboveground biomass, belowground biomass, dead wood, litter, soil organic carbon, harvested wood products in use, and harvested wood products in landfills. The estimated carbon dioxide (CO_2) flux from each of these forest components was derived from U.S. forest inventory data, using methodologies that are consistent with *LULUCF Good Practice Guidance* (IPCC 2003) and the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). In addition, this year, according to the new *LULUCF Good Practice Guidance* (IPCC 2003), N_2O emissions from fertilized forest soils are accounted for utilizing a default methodology.

Croplands remaining croplands emission estimates are a reflection of the changes in agricultural soil carbon stocks on both cropland and grazing land since the necessary datasets were not available to separate cropland and grassland fluxes. Changes in agricultural soil carbon stocks include mineral and organic soil carbon stock changes due to use and management of cropland and grazing land, and emissions of CO_2 due to the application of crushed limestone and dolomite to agricultural soils (i.e., soil liming). The methods used to estimate all three components of flux in agricultural soil carbon stocks are consistent with the *Revised 1996 IPCC Guidelines* and the *LULUCF Good Practice Guidance (IPCC 2003)*.

Fluxes resulting from settlements remaining settlements include landfilled yard trimmings and food scraps, urban trees, and soil N_2O emissions from fertilization. Changes in yard trimming and food scrap carbon stocks in landfills are estimated using analysis of life-cycle greenhouse gas emissions and sinks associated with solid waste management (EPA 1998). Changes in carbon stocks in urban trees are estimated based on field measurements in ten U.S. cities and data on national urban tree cover, using a methodology consistent with the *LULUCF Good Practice Guidance* (IPCC 2003). Finally, this year, according to the new *LULUCF Good Practice Guidance* (IPCC 2003), N_2O emissions from fertilized settlement soils are accounted for according to a default methodology. Note that the chapter title "Land-Use Change and Forestry" has been used here to maintain consistency with the IPCC reporting structure for national greenhouse gas inventories; however, the chapter covers land-use activities, in addition to land-use change and forestry activities. Therefore, except in table titles, the term "land use, land-use change, and forestry" will be used in the remainder of this chapter.

[1] The term "flux" is used here to encompass both emissions of greenhouse gases to the atmosphere, and removal of carbon from the atmosphere. Removal of carbon from the atmosphere is also referred to as "carbon sequestration."

Unlike the assessments in other sectors, which are based on annual activity data, the flux estimates in this chapter, with the exception of those from wood products, urban trees, liming, and settlement and forest N_2O emissions, are based on periodic activity data in the form of forest, land-use, and municipal solid waste surveys. Carbon dioxide fluxes from forest carbon stocks (except the wood product components) and from agricultural soils (except the liming component) are calculated on an average annual basis from data collected in intervals ranging from 1 to 10 years. The resulting annual averages are applied to years between surveys. Because state surveys are collected at different times, using this data structure, the estimated CO_2 fluxes from forest carbon stocks differ at the national level from year to year. Agricultural soil carbon flux calculations are constant over multi-year intervals, with large discontinuities between intervals; however, fluxes after 1997 are inconsistent due to the method of accounting for the application of manure and sewage sludge amendments to mineral soils after 1997. For the landfilled yard trimmings and food scraps source, periodic solid waste survey data were interpolated so that annual storage estimates could be derived. In addition, because the most recent national forest, land-use, and municipal solid waste surveys were completed prior to 2003, the estimates of CO_2 flux from forests, agricultural soils, and landfilled yard trimmings and food scraps are based in part on modeled projections or extrapolation. Carbon dioxide flux from urban trees is based on neither annual data nor periodic survey data, but instead on data collected over the period 1990 through 1999. This flux has been applied to the entire time series.

Land use, land-use change, and forestry activities in 2003 resulted in a net carbon sequestration of 828 Tg CO_2 Eq. (226 Tg C) (Table 7-1and Table 7-2). This represents an offset of approximately 14 percent of total U.S. CO_2 emissions. Total land use, land-use change, and forestry net carbon sequestration declined by approximately 21 percent between 1990 and 2003. This decline was primarily due to a decline in the rate of net carbon accumulation in forest carbon stocks. Annual carbon accumulation in landfilled yard trimmings and food scraps also slowed over this period, as did annual carbon accumulation in agricultural soils. As described above, the constant rate of carbon accumulation in urban trees is a reflection of limited underlying data (i.e., this rate represents an average for 1990 through 1999).

Table 7-1: Net CO_2 Flux from Land-Use Change and Forestry (Tg CO_2 Eq.)

Sink Category	1990	1997	1998	1999	2000	2001	2002	2003
Forest Land Remaining Forest Land	**(949.3)**	**(851.0)**	**(805.5)**	**(751.7)**	**(747.9)**	**(750.9)**	**(751.5)**	**(752.7)**
Changes in Forest Carbon Stocks	(949.3)	(851.0)	(805.5)	(751.7)	(747.9)	(750.9)	(751.5)	(752.7)
Cropland Remaining Cropland	**(8.1)**	**(7.4)**	**(4.3)**	**(4.3)**	**(5.7)**	**(7.1)**	**(6.2)**	**(6.6)**
Changes in Agricultural Soil Carbon Stocks	(8.1)	(7.4)	(4.3)	(4.3)	(5.7)	(7.1)	(6.2)	(6.6)
Settlements Remaining Settlements	**(84.7)**	**(71.6)**	**(71.2)**	**(70.0)**	**(68.9)**	**(68.9)**	**(68.8)**	**(68.7)**
Urban Trees	(58.7)	(58.7)	(58.7)	(58.7)	(58.7)	(58.7)	(58.7)	(58.7)
Landfilled Yard Trimmings and Food Scraps	(26.0)	(12.9)	(12.5)	(11.4)	(10.2)	(10.3)	(10.2)	(10.1)
Total	**(1042.0)**	**(930.0)**	**(881.0)**	**(826.1)**	**(822.4)**	**(826.9)**	**(826.5)**	**(828.0)**

Note: Parentheses indicate net sequestration. Totals may not sum due to independent rounding.

Table 7-2: Net CO_2 Flux from Land-Use Change and Forestry (Tg C)

Sink Category	1990	1997	1998	1999	2000	2001	2002	2003
Forest Land Remaining Forest Land	**(259)**	**(232)**	**(220)**	**(205)**	**(204)**	**(205)**	**(205)**	**(205)**
Changes in Forest Carbon Stocks	(259)	(232)	(220)	(205)	(204)	(205)	(205)	(205)
Cropland Remaining Cropland	**(2)**	**(2)**	**(1)**	**(1)**	**(2)**	**(2)**	**(2)**	**(2)**
Changes in Agricultural Soil Carbon Stocks	(2)	(2)	(1)	(1)	(2)	(2)	(2)	(2)
Settlements Remaining Settlements	**(23)**	**(20)**	**(19)**	**(19)**	**(19)**	**(19)**	**(19)**	**(19)**
Urban Trees	(16)	(16)	(16)	(16)	(16)	(16)	(16)	(16)
Landfilled Yard Trimmings and Food Scraps	(7)	(4)	(3)	(3)	(3)	(3)	(3)	(3)
Total	**(284)**	**(254)**	**(240)**	**(225)**	**(224)**	**(226)**	**(225)**	**(226)**

Note: 1 Tg C = 1 teragram carbon = 1 million metric tons carbon. Parentheses indicate net sequestration. Totals may not sum due to independent rounding.

Land use, land-use change, and forestry activities in 2003 resulted in a net flux of 6.4 Tg CO_2 Eq. of N_2O (20.7 Gg) (Table 7-3 and Table 7-4). Total N_2O emissions from the application of fertilizers to forests and settlements increased by approximately 14 percent between 1990 and 2003.

Table 7-3: Net N_2O Emissions from Land-Use Change and Forestry (Tg CO_2 Eq.)

Sink Category	1990		1997	1998	1999	2000	2001	2002	2003
Forest Land Remaining Forest Land	**0.1**		**0.3**	**0.4**	**0.5**	**0.4**	**0.4**	**0.4**	**0.4**
N_2O Fluxes from Soils	0.1		0.3	0.4	0.5	0.4	0.4	0.4	0.4
Settlements Remaining Settlements	**5.5**		**6.1**	**6.1**	**6.2**	**6.0**	**5.8**	**6.0**	**6.0**
N_2O Fluxes from Soils	5.5		6.1	6.1	6.2	6.0	5.8	6.0	6.0
Total	**5.6**		**6.4**	**6.5**	**6.6**	**6.3**	**6.2**	**6.4**	**6.4**

Note: Totals may not sum due to independent rounding.

Table 7-4: Net N_2O Emissions from Land-Use Change and Forestry (Gg)

Sink Category	1990		1997	1998	1999	2000	2001	2002	2003
Forest Land Remaining Forest Land	**0.2**		**1.0**	**1.1**	**1.5**	**1.1**	**1.3**	**1.3**	**1.3**
N_2O Fluxes from Soils	0.2		1.0	1.1	1.5	1.1	1.3	1.3	1.3
Settlements Remaining Settlements	**17.9**		**19.8**	**19.8**	**19.9**	**19.3**	**18.7**	**19.4**	**19.4**
N_2O Fluxes from Soils	17.9		19.8	19.8	19.9	19.3	18.7	19.4	19.4
Total	**18.1**		**20.7**	**20.9**	**21.4**	**20.4**	**20.0**	**20.7**	**20.7**

Note: Totals may not sum due to independent rounding.

7.1. Forest Land Remaining Forest Land

Changes in Forest Carbon Stocks (IPCC Source Category 5A1)

For estimating carbon (C) stocks or stock change (flux), C in forest ecosystems can be divided into the following five storage pools (IPCC 2003):

- Aboveground biomass, all living biomass above the soil including stem, stump, branches, bark, seeds, and foliage. This category includes live understory.
- Belowground biomass, all living biomass of coarse living roots greater than 2 mm diameter.
- Dead wood, including all non-living woody biomass either standing, lying on the ground (but not including litter), or in the soil.
- Litter, including the litter, fumic, and humic layers, and all non-living biomass with a diameter less than 7.5 cm at transect intersection, lying on the ground.
- Soil organic carbon (SOC), including all organic material in soil to a depth of 1 meter but excluding the coarse roots of the above pools.

In addition, there are two harvested wood pools also necessary for estimating C flux, which are:

- Harvested wood products in use.
- Harvested wood products in landfills.

Carbon is continuously cycled among these storage pools and between forest ecosystems and the atmosphere as a result of biological processes in forests (e.g., photosynthesis, growth, mortality, decomposition, and disturbances such as fires or pest outbreaks) and anthropogenic activities (e.g., harvesting, thinning, clearing, and replanting). As trees photosynthesize and grow, C is removed from the atmosphere and stored in living tree biomass. As trees age, they continue to accumulate C until they reach maturity, at which point they store a relatively constant amount of C. As trees die and otherwise deposit litter and debris on the forest floor, C is released to the atmosphere or transferred to the soil by organisms that facilitate decomposition.

The net change in forest C is not equivalent to the net flux between forests and the atmosphere because timber harvests do not cause an immediate flux of C to the atmosphere. Instead, harvesting transfers C to a "product pool." Once in a product pool, the C is emitted over time as CO_2 when the wood product combusts or decays. The rate of

emission varies considerably among different product pools. For example, if timber is harvested to produce energy, combustion releases C immediately. Conversely, if timber is harvested and used as lumber in a house, it may be many decades or even centuries before the lumber decays and C is released to the atmosphere. If wood products are disposed of in landfills, the C contained in the wood may be released many years or decades later, or may be stored almost permanently in the landfill.

This section quantifies the net changes in C stocks in the five forest C pools and two harvested wood pools. The net change in stocks for each pool is estimated, and then the changes in stocks are summed over all pools to estimate total net flux. Thus, the focus on C implies that all C-based greenhouse gases are included, and the focus on stock change suggests that specific ecosystem fluxes are not separately itemized in this report. Disturbances from forest fires and pest outbreaks are implicitly included in the net changes. For instance, an inventory conducted after fire counts only trees left. The change between inventories thus counts the carbon changes due to fires; however, it may not be possible to attribute the changes to the disturbance specifically. The IPCC *LULUCF Good Practice Guidance* (IPCC 2003) recommends reporting C stocks according to several land use types and conversions, specifically forest land remaining forest land, nonforest land becoming forest, and forest becoming non-forest. Currently, consistent datasets are not available for the entire United States to allow results to be partitioned in this way. Instead, net changes in all forest-related land, including non-forest land converted to forest and forests converted to non-forest are reported here.

Forest C storage pools, and the flows between them via emissions, sequestration, and transfers, are shown in Figure 7-1. In the figure, boxes represent forest C storage pools and arrows represent flows between storage pools or between storage pools and the atmosphere. Note that the boxes are not identical to the storage pools identified in this chapter. The storage pools identified in this chapter have been altered in this graphic to better illustrate the processes that result in transfers of C from one pool to another, and emissions to the atmosphere as well as uptake from the atmosphere.

Figure 7-1: Forest Sector Carbon Pools and Flows.

Note: Boxes represent forest C storage pools and arrows represent flows between storage pools or between storage pools and the atmosphere.

Approximately 33 percent (303 million hectares) of the U.S. land area is forested (Smith et al. 2004b). From the early 1970s to the early 1980s, forest land declined by approximately 2.4 million hectares. During the 1980s and 1990s, forest area increased by about 3.7 million hectares. These net changes in forest area represent average annual fluctuations of only about 0.1 percent. Given the low rate of change in U.S. forest land area, the major influences on the current net C flux from forest land are management activities and the ongoing impacts of previous land-use changes. These activities affect the net flux of C by altering the amount of C stored in forest ecosystems. For example, intensified management of forests can increase both the rate of growth and the eventual biomass density[2] of the forest, thereby increasing the uptake of C. Harvesting forests removes much of the aboveground C, but trees can grow on this area again and sequester C. The reversion of cropland to forest land increases C storage in biomass, forest floor, and soils. The net effects of forest management and the effects of land-use change involving forest lands are captured in the estimates of C stocks and fluxes presented in this chapter.

In the United States, improved forest management practices, the regeneration of previously cleared forest areas, as well as timber harvesting and use have resulted in net uptake (i.e., net sequestration) of C each year from 1990

[2] The term "biomass density" refers to the mass of vegetation per unit area. It is usually measured on a dry-weight basis. Dry biomass is about 50 percent carbon by weight.

through 2003. Due to improvements in U.S. agricultural productivity, the rate of forest clearing for crop cultivation and pasture slowed in the late 19th century, and by 1920 this practice had all but ceased. As farming expanded in the Midwest and West, large areas of previously cultivated land in the East were taken out of crop production, primarily between 1920 and 1950, and were allowed to revert to forests or were actively reforested. The impacts of these land-use changes still affect C fluxes from forests in the East. In addition, C fluxes from eastern forests have been affected by a trend toward managed growth on private land. Collectively, these changes have nearly doubled the biomass density in eastern forests since the early 1950s. More recently, the 1970s and 1980s saw a resurgence of federally-sponsored forest management programs (e.g., the Forestry Incentive Program) and soil conservation programs (e.g., the Conservation Reserve Program), which have focused on tree planting, improving timber management activities, combating soil erosion, and converting marginal cropland to forests. In addition to forest regeneration and management, forest harvests have also affected net C fluxes. Because most of the timber harvested from U.S. forests is used in wood products, and many discarded wood products are disposed of in landfills rather than by incineration, significant quantities of C in harvested wood are transferred to long-term storage pools rather than being released rapidly to the atmosphere (Skog and Nicholson 1998). The size of these long-term C storage pools has increased during the last century.

Changes in C stocks in U.S. forests and harvested wood were estimated to account for an average annual net sequestration of 832 Tg CO_2 Eq. (227 Tg C) over the period 1990 through 2003 (Table 7-5, Table 7-6, and Figure 7-2). In addition to the net accumulation of C in harvested wood pools, sequestration is a reflection of net forest growth and increasing forest area over this period, particularly before 1997. The increase in forest sequestration is due more to an increasing C density per area than to the increase in area of forestland. Forestland in the conterminous United States was approximately 246, 250, and 251 million hectares for 1987, 1997, and 2002, respectively, only a 2 percent increase over the period (Smith et al. 2004b). Continuous, regular annual surveys are not available over the period for each state; therefore, estimates for non-survey years were derived by interpolation between known data points. Survey years vary from state to state. National estimates are a composite of individual state surveys. Total sequestration declined by 21 percent between 1990 and 2003. This decline was primarily due to a decline in the estimated rate of sequestration in forest soils. Inventory derived estimates of soil C stocks are based solely on forest area and type. Thus, changes in soil C over time are directly the result of changes in total forest area or changes in forest type from forest inventory data.

Table 7-5. Net Annual Changes in Carbon Stocks (Tg CO_2 Eq. yr^{-1}) in Forest and Harvested Wood Pools

Carbon Pool	1990	1997	1998	1999	2000	2001	2002	2003
Forest	**(739)**	**(638)**	**(599)**	**(537)**	**(537)**	**(537)**	**(537)**	**(537)**
Aboveground Biomass	(396)	(457)	(437)	(400)	(400)	(400)	(400)	(400)
Belowground Biomass	(77)	(89)	(85)	(78)	(78)	(78)	(78)	(78)
Dead Wood	(74)	(53)	(51)	(45)	(45)	(45)	(45)	(45)
Litter	(67)	(31)	(28)	(26)	(26)	(26)	(26)	(26)
Soil Organic Carbon	(125)	(8)	1	12	12	12	12	12
Harvested Wood	**(210)**	**(213)**	**(206)**	**(215)**	**(211)**	**(214)**	**(214)**	**(216)**
Wood Products	(48)	(58)	(52)	(62)	(59)	(59)	(59)	(60)
Landfilled Wood	(162)	(155)	(154)	(153)	(152)	(155)	(155)	(155)
Total Net Flux	**(949)**	**(851)**	**(806)**	**(752)**	**(748)**	**(751)**	**(751)**	**(753)**

Note: Parentheses indicate net C sequestration (i.e., a net removal of C from the atmosphere). Total net flux is an estimate of the actual net flux between the total forest C pool and the atmosphere. Forest estimates are based on interpolation and extrapolation of inventory data as described in the text and in Annex 3.12. Harvested wood estimates are based on results from annual surveys and models. Totals may not sum due to independent rounding.

Table 7-6. Net Annual Changes in Carbon Stocks (Tg C yr^{-1}) in Forest and Harvested Wood Pools

Carbon Pool	1990	1997	1998	1999	2000	2001	2002	2003
Forest	**(202)**	**(174)**	**(163)**	**(146)**	**(146)**	**(146)**	**(146)**	**(146)**
Aboveground Biomass	(108)	(125)	(119)	(109)	(109)	(109)	(109)	(109)
Belowground Biomass	(21)	(24)	(23)	(21)	(21)	(21)	(21)	(21)
Dead Wood	(20)	(14)	(14)	(12)	(12)	(12)	(12)	(12)
Litter	(18)	(9)	(8)	(7)	(7)	(7)	(7)	(7)

Soil Organic Carbon	(34)	(2)	0	3	3	3	3	3
Harvested Wood	**(57)**	**(58)**	**(56)**	**(59)**	**(57)**	**(58)**	**(58)**	**(59)**
Wood Products	(13)	(16)	(14)	(17)	(16)	(16)	(16)	(16)
Landfilled Wood	(44)	(42)	(42)	(42)	(41)	(42)	(42)	(42)
Total Net Flux	**(259)**	**(232)**	**(220)**	**(205)**	**(204)**	**(205)**	**(205)**	**(205)**

Note: Parentheses indicate net C sequestration (i.e., a net removal of C from the atmosphere). Total net flux is an estimate of the actual net flux between the total forest C pool and the atmosphere. Forest estimates are based on interpolation and extrapolation of inventory data as described in the text and in Annex 3.12. Harvested wood estimates are based on results from annual surveys and models. Totals may not sum due to independent rounding.

Stock estimates for forest and harvested wood C storage pools are presented in Table 7-7. Together, the aboveground live and forest soil pools account for a large proportion of total forest C stocks. C stocks in all non-soil pools increased over time. Therefore, C sequestration was greater than C emissions from forests, as discussed above. Figure 7-3 shows the average carbon density in forests by state, estimated for 2004.

Table 7-7. Carbon Stocks (Tg C) in Forest and Harvested Wood Pools

Carbon Pool	1990	1997	1998	1999	2000	2001	2002	2003	2004
Forest	**39,498**	**40,812**	**40,986**	**41,149**	**41,296**	**41,442**	**41,589**	**41,735**	**41,882**
Aboveground Biomass	14,114	14,928	15,053	15,172	15,281	15,390	15,499	15,608	15,717
Belowground Biomass	2,805	2,963	2,987	3,011	3,032	3,053	3,074	3,095	3,117
Dead Wood	2,444	2,572	2,587	2,600	2,613	2,625	2,638	2,650	2,662
Litter	4,496	4,598	4,606	4,614	4,621	4,628	4,636	4,643	4,650
Soil Organic Carbon	15,640	15,750	15,752	15,752	15,749	15,745	15,742	15,738	15,735
Harvested Wood	**1,915**	**2,307**	**2,365**	**2,421**	**2,480**	**2,537**	**2,595**	**2,654**	**2,713**
Wood Products	1,134	1,232	1,248	1,262	1,279	1,295	1,311	1,327	1,344
Landfilled Wood	781	1,074	1,117	1,159	1,200	1,242	1,284	1,327	1,369
Total Carbon Stock	**41,414**	**43,119**	**43,351**	**43,570**	**43,775**	**43,979**	**44,184**	**44,389**	**44,594**

Note: Forest C stocks do not include forest stocks in Alaska, Hawaii, or U.S. territories, or trees on non-forest land (e.g., urban trees). Wood product stocks include exports, even if the logs are processed in other countries, and exclude imports. Forest estimates are based on interpolation and extrapolation of inventory data as described in the text and in Annex 3.12. Harvested wood estimates are based on results from annual surveys and models. Totals may not sum due to independent rounding. Inventories are assumed to represent stocks as of January 1 of the inventory year. Flux is the net annual change in stock. Thus, an estimate of flux for 2003 requires estimates of C stocks for 2003 and 2004.

Figure 7-2: Estimates of Net Annual Changes in Carbon Stocks for Major Carbon Pools (Tg C yr^{-1})

Note: Estimates for harvested wood are based on the same methodology and data as the previous U.S. Inventory (EPA 2004). Estimates for all pools are based on measured forest inventory data as described in the text. Total Net includes all forest pools: biomass, dead wood, litter, forest soils, wood products, and landfilled wood.

Figure 7-3: Average Carbon Density in the Forest Tree Pool in the Conterminous U.S. During 2004.

Methodology

The methodology described herein is consistent with *LULUCF Good Practice Guidance* (IPCC 2003) and the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). Estimates of net C flux from Land-Use Change and Forestry, including all pools except harvested wood, were derived from periodic and annualized inventories of

forest stocks. Net changes in C stocks were interpolated between survey years. Carbon emissions from harvested wood were determined by accounting for the variable rate of decay of harvested wood according to its disposition (e.g., product pool, landfill, combustion).[3] Different data sources were used to estimate the C stocks and stock change in (1) forests (aboveground and belowground biomass, dead wood, and litter), (2) forest soils, and (3) harvested wood products. Therefore, these pools are described separately below.

Live Biomass, Dead Wood, and Litter Carbon

The estimates of non-soil forest C stocks are based on data derived from forest inventory surveys. Forest survey data were obtained from the USDA Forest Service, Forest Inventory and Analysis (FIA) program (Frayer and Furnival 1999, Smith et al. 2001). Surveys provide estimates of the merchantable volume of wood and other variables that are used to estimate C stocks. Estimates of temporal change such as growth, mortality, harvests, or area change are derived from repeated surveys, which were conducted every 5 to 14 years, depending on the state. Historically, the FIA program did not conduct detailed surveys of all forest land, but instead focused on land capable of supporting timber production (timberland[4]). However, over time individual state surveys gradually started to include reserved and less productive forest lands. The C stock estimates provided here include all forest land, see Annex 3.12 for discussion of how past data gaps on these lands were filled.

Temporal and spatial gaps in surveys were addressed with the new national plot design and annualized sampling (Miles et al. 2001, Alerich et al. 2004), which were recently introduced by FIA. Annualized sampling means that a portion of plots throughout each state is sampled each year, with the goal of measuring all plots once each 5 years. Sampling is designed such that partial inventory cycles provide usable, unbiased samples of forest inventory. Thus, many states have relatively recent partial inventories, yet not all states are currently surveyed this way. All annualized surveys initiated since 1998 have followed the new national plot design for all forestlands, including reserved and less productive lands. Inventories are assumed to represent stocks as of January 1 of the inventory year.

For each periodic or annualized inventory in each state, each C pool was estimated using coefficients from the FORCARB2 model (Birdsey and Heath 1995, Birdsey and Heath 2001, Heath et al. 2003, Smith et al. 2004a). Estimates of C stocks made by the FORCARB2 coefficients at the plot level are organized somewhat differently than the standard IPCC pools reported in Table 7-7. However, the estimators are compatible with reorganizing the pools following IPCC *LULUCF Good Practice Guidance* (2003). For example, the biomass pools here include the FORCARB2 pools of live trees and understory vegetation, each of which are divided into aboveground versus belowground portions. Calculations for the tree portion of the aboveground biomass C pool were made using volume-to-biomass conversion factors for different types of forests as presented in Smith et al. (2003). Biomass was converted to C mass by dividing by two because dry biomass is approximately 50 percent C (IPCC/UNEP/OECD/IEA 1997). The other portion of aboveground biomass, live understory C, was estimated from inventory data using tables presented in Birdsey (1996). Litter C was estimated from inventory data using the equations presented in Smith and Heath (2002). Down dead wood was estimated using a FORCARB2 simulation and U.S. forest statistics (Smith et al. 2001).

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[3] The wood product stock and flux estimates presented here use the production approach, meaning that they do not account for C stored in imported wood products, but do include C stored in exports, even if the logs are processed in other countries. This approach is used because it follows the precedent established in previous reports (Heath et al. 1996).

[4] Forest land in the U.S. includes land that is at least 10 percent stocked with trees of any size. Timberland is the most productive type of forest land, which is on unreserved land and is producing or capable of producing crops of industrial wood. Productivity is at a minimum rate of 20 cubic feet of industrial wood per acre per year. The remaining portion of forest land is classified as either reserved forest land, which is forest land withdrawn from timber use by statute or regulation, or other forest land, which includes less productive forests on which timber is growing at a rate less than 20 cubic feet per acre per year. In 2002, there were about 199 million hectares of timberland in the conterminous U.S., which represented 79 percent of all forest lands over the same area (Smith et al. 2004b).

Forest Soil Carbon

Estimates of soil organic carbon stocks are based solely on forest area and on average soil C density for each broad forest type group. Thus, any changes in soil C stocks are due to changes in total forest area or the distribution of forest types within that area. Estimates of the organic C content of soils are based on the national STATSGO spatial database (USDA 1991) and follow methods of Amichev and Galbraith (2004). These data were overlaid with FIA survey data to estimate soil C on forest lands by broad forest type group.

Forest Carbon Stocks and Fluxes

The overall approach for determining forest C stock change was to estimate forest C stocks based on data from two forest surveys conducted several years apart. Carbon stocks were calculated separately for each state based on inventories available since 1990 and for the most recent inventory prior to 1990. Thus, the number of separate stock estimates for each state was one less than the number of available inventories. For each pool in each state in each year, C stocks were estimated by linear interpolation between survey years. Similarly, fluxes were estimated for each pool in each state by dividing the difference between two successive stocks by the number of intervening years between surveys. Note that inventories are assumed to represent stocks as of January 1 of the inventory year; thus, stocks in 1989 and 1993 can be used to estimate flux for 1989 through 1992, for example. Stocks and fluxes since the most recent survey were based on extrapolating estimates from the last two surveys. C stock and flux estimates for each pool were summed over all states to form estimates for the conterminous United States. Data sources and methods for estimating individual C pools are described more fully in Annex 3.12.

Harvested Wood Carbon

Estimates of C stock changes in wood products and wood discarded in landfills were based on the methods described by Skog and Nicholson (1998). Carbon stocks in wood products in use and wood products stored in landfills were estimated from 1910 onward based on historical data from the USDA Forest Service (USDA 1964, Ulrich 1989, Howard 2001), and historical data as implemented in the framework underlying the North American Pulp and Paper (NAPAP, Ince 1994) and the Timber Assessment Market and the Aggregate Timberland Assessment System Timber Inventory models (TAMM/ATLAS, Haynes 2003, Mills and Kincaid 1992). Beginning with data on annual wood and paper production, the fate of C in harvested wood was tracked for each year from 1910 through 2003, and included the change in C stocks in wood products, the change in C in landfills, and the amount of C emitted to the atmosphere (CO_2 and CH_4) both with and without energy recovery. To account for imports and exports, the production approach was used, meaning that C in exported wood was counted as if it remained in the United States, and C in imported wood was not counted.

Uncertainty

The forest survey data that underlie the forest C estimates are based on a statistical sample designed to represent the wide variety of growth conditions present over large territories. However, forest survey data that are currently available generally exclude timber stocks on most forest land in Alaska, Hawaii, and U.S. territories. For this reason, estimates have been developed only for the conterminous United States. Within the conterminous United States, the USDA Forest Service mandates that forest area data are accurate within 3 percent at the 67 percent confidence level (one standard error) per 405,000 ha of forest land (Miles et al. 2001). For larger areas, the uncertainty in area is concomitantly smaller. For volume data, the accuracy is targeted to be 5 percent for each 28,300 m^3 at the same confidence level. An analysis of uncertainty in growing stock volume data for timber producing lands was undertaken for five states: Florida, Georgia, North Carolina, South Carolina, and Virginia (Phillips et al. 2000). Nearly all of the uncertainty was found to be due to sampling rather than the regression equations used to estimate volume from tree height and diameter. Standard errors for growing stock volume ranged from 1 to 2 percent for individual states and less than 1 percent for the 5-state region. However, the total standard error for the change in growing stock volume was estimated to be 12 to 139 percent for individual states, and 20 percent for the 5-state region. The high relative uncertainty for growing stock volume change in some states was due to small net changes in growing stock volume. However, the uncertainty in volume change may be smaller than was found in this study because estimates from samples taken at different times on permanent survey plots are

correlated, and such correlation reduces the uncertainty in estimates of changes in volume or C over time (Smith and Heath 2000). Based on these accuracy guidelines and these results for the Southeastern United States, forest area and volume data for the conterminous United States are expected to be reasonably accurate, although estimates of small changes in growing stock volume may have substantial uncertainty.

In addition to uncertainty in growing stock volume, there is uncertainty associated with the estimates of C stocks in other ecosystem pools. Estimates for these pools are derived from extrapolations of site-specific studies to all forest land since survey data on these pools are not generally available. Such extrapolation introduces uncertainty because available studies may not adequately represent regional or national averages. Uncertainty may also arise due to (1) modeling errors, for example relying on coefficients or relationships that are not well known, and (2) errors in converting estimates from one reporting unit to another (Birdsey and Heath 1995). An important source of uncertainty is that there is little consensus from available data sets on the effect of land use change and forest management activities (such as harvest) on soil C stocks. For example, while Johnson and Curtis (2001) found little or no net change in soil C following harvest, on average, across a number of studies, many of the individual studies did exhibit differences. Heath and Smith (2000b) noted that the experimental design in a number of soil studies limited their usefulness for determining effects of harvesting on soil C. Because soil C stocks are large, estimates need to be very precise, since even small relative changes in soil C sum to large differences when integrated over large areas. The soil C stock and stock change estimates presented herein are based on the assumption that soil C density for each broad forest type group stays constant over time. As more information becomes available, the effects of land use and of changes in land use and forest management will be better accounted for in estimates of soil C (see "Planned Improvements," below).

Recent studies have begun to quantify the uncertainty in national-level forest C budgets based on the methods adopted here. Smith and Heath (2000) and Heath and Smith (2000a) report on an uncertainty analysis they conducted on C sequestration in privately owned timberlands throughout the conterminous United States. These studies are not directly comparable to the estimates in this chapter because they used an older version of the FORCARB model and are based on older data. However, the relative magnitudes of the uncertainties are informative. For the period 1990 through 1999, the true mean C flux was estimated to be within 15 percent of the reported mean at the 80 percent confidence level. The corresponding true mean C stock estimate for 2000 was within approximately 5 percent of the reported mean value at the 80 percent confidence level. The relatively greater uncertainty in flux estimates compared to stock estimates is roughly similar to that found for estimates of growing stock volume discussed above (Phillips et al. 2000). In both analyses, there are greater relative uncertainties associated with smaller estimates of flux. Uncertainty in the estimates presented in this inventory may be greater than those presented by Heath and Smith (2000a) for several reasons. Most importantly, their analysis did not include uncertainty in growing stock volume data or uncertainties in stocks and fluxes of C from harvested wood.

The uncertainty analysis was performed using the IPCC-recommended Tier 2 uncertainty estimation methodology, Monte Carlo Simulation technique. The results of the Tier 2 quantitative uncertainty analysis are summarized in Table 7-8. The 2003 flux estimate for forest C stocks is estimated to be between (1,120.5) and (383.5) Tg CO_2 Eq. at a 95 percent confidence level (or 19 of 20 Monte Carlo Stochastic Simulations). This indicates a range of 49 percent below to 49 percent above the 2003 flux estimate of (752.7) Tg CO_2 Eq.

Table 7-8: Tier 2 Quantitative Uncertainty Estimates for CO_2 Net Flux from Forest Land Remaining Forest Land: Changes in Forest Carbon Stocks (Tg CO_2 Eq. and Percent)

Source	Gas	2003 Flux Estimate (Tg CO_2 Eq.)	Uncertainty Range Relative to 2003 Estimate[a] (Tg CO_2 Eq.)		Uncertainty Range Relative to 2003 Estimate[a] (%)	
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Forests Remaining Forests: Changes in Forest Carbon Stocks	CO_2	(752.7)	(1,120.5)	(383.5)	-49%	+49%

[a]Range of flux estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

QA/QC and Verification

As discussed above, the USDA Forest Service Forest Inventory and Analysis program has conducted consistent forest surveys based on extensive statistically-based sampling of most of the forest land in the conterminous United States since 1952. The main purpose of the Forest Inventory and Analysis program has been to estimate areas, volume of growing stock, and timber products output and utilization factors. The Forest Inventory and Analysis program includes numerous quality assurance and quality control procedures, including calibration among field crews, duplicate surveys of some plots, and systematic checking of recorded data. Because of the statistically-based sampling, the large number of survey plots, and the quality of the data, the survey databases developed by the Forest Inventory and Analysis program form a strong foundation for C stock estimates. Field sampling protocols, summary data, and detailed inventory databases are archived and are publicly available on the Internet (FIA Homepage).

Many key calculations for estimating current forest C stocks based on FIA data are based on coefficients from the FORCARB2 model (see additional discussion in the Methods section above and in Annex 3.12). The model has been used for many years to produce national assessments of forest C stocks and stock changes. General quality control procedures were used in performing calculations to estimate C stocks based on survey data. For example, the derived C datasets, which include inventory variables such as areas and volumes, were compared with standard inventory summaries such as Resources Planning Act (RPA) Forest Resource Tables or selected population estimates generated from the FIADB, which are available at an FIA Internet site (FIA Database Retrieval System). Agreement between the C datasets and the original inventories is important to verify accuracy of the data used. Forest Inventory and Analysis data and some model projections are given in English units, but C stock estimates were developed using metric units. To avoid unit conversion errors, a standard conversion table in electronic form was used (Appendix B of Smith et al. 2001). Finally, C stock estimates were compared with previous inventory report estimates to assure that any differences could be explained by either new data or revised calculation methods (see the "Recalculations" discussion below).

Recalculations Discussion

The overall scheme for developing annualized estimates of C stocks based on the individual state surveys is similar to that presented in the previous Inventory (EPA 2003). Methods for estimating soil organic carbon are new for the current Inventory; differences are in the interpretation of STATSGO data and their relationship with FIA survey data as described by Amichev and Galbraith (2004)—see Annex 3.12 for additional information. Similar to that reported in the previous Inventory, estimates of forest C stocks and fluxes are based on forest inventory data from individual states rather than regions, and the data collected in states were assigned an average survey inventory plot date rather than simply assigned the year for which the database was compiled. However, the selection and compilation of survey data was implemented differently for the current Inventory, and there were some important differences in the underlying data.

Three differences in methods can affect the non-soil forest C estimates. First, the selection of the datasets representing individual state surveys was independent of last year's selections. Both RPA and the newer FIADB datasets were considered, whereas last year only RPA data were used. The RPA data represent specific compilations of survey data and include some older data not currently available in the FIADB. Using both ensured the most recent data were used, yet older data were available as needed. Inventory data—even older surveys—are occasionally modified so that RPA and FIADB data of ostensibly the same survey may have some slight differences that can affect the C estimates. This is likely to have a very minor effect on recalculation of C stocks. Another minor change in method is that fluxes were separately determined from the original survey data for each state for each pool; in contrast, last year stocks were interpolated and summed to a national total for each year before flux was calculated. Finally, separate stocks and fluxes were determined for National Forest lands where, in the past, independent surveys were conducted at distinctly different times.

Pool definitions have changed for the current Inventory, as suggested by IPCC *LULUCF Good Practice Guidance* (IPCC 2003). In previous Inventories, the pools were trees, understory, forest floor, down dead wood, and forest soils. The forest soil pool is now soil organic carbon; forest floor is called litter. The previous tree pool included both above- and belowground biomass and mass of standing dead trees. The mass of standing dead trees was added

to the down dead wood pool and is called dead wood. The remainder of the tree pool, live biomass, as well as the understory pool, was split into above- and belowground portions. The aboveground tree and understory pools were summed into the aboveground biomass pool; the belowground portions of these pools were added to create the belowground biomass pools.

Two changes in the use of data are also likely to affect the recalculation of C. The equations used to estimate tree C from forest inventory data have been revised slightly; the net effect is that total tree C (live plus standing dead trees, which are part of both the biomass and dead wood pools as summarized here) calculated for the 2002 RPA database (Smith et al. 2004b) was 0.3 percent greater with the new set of equations relative to those used last year. Perhaps the largest effect on C recalculations is that for the previous Inventory the final C stocks were modeled. This year, however, values are simply extrapolated. The principal reason for eliminating the projections was the difficulty in establishing projections consistent with the available forest inventory data.

Overall, these changes resulted in an average annual increase of 103.3 Tg CO_2 Eq. (16 percent) in forest carbon stocks for the period 1990 through 2002.

Planned Improvements

The Forest Inventory and Analysis program has adopted a new annualized design, such that a portion of each state will be surveyed each year (Gillespie 1999). The annualized survey also includes measuring attributes that are needed to estimate C in various pools, such as soil C and forest floor C, on a subset of the plots. During the next several years, the use of annual data, including new data on soil and forest floor C stocks, and new data on non-timberlands, will improve the precision and accuracy of estimates of forest C stocks and fluxes.

As more information becomes available about historical land use, the ongoing effects of changes in land use and forest management will be better accounted for in estimates of soil C (Birdsey and Lewis 2003). Currently, soil C estimates are based on the assumption that soil C density depends only on broad forest type group, not on land use history. However, many forests in the Eastern United States are re-growing on abandoned agricultural land. During such regrowth, soil and forest floor C stocks often increase substantially over many years or even decades, especially on highly eroded agricultural land. In addition, with deforestation, soil C stocks often decrease over many years. A new methodology is being developed to account for these changes in soil C over time. This methodology includes estimates of area changes among land uses (especially forest and agriculture), estimates of the rate of soil C stock gain with afforestation, and estimates of the rate of soil C stock loss with deforestation over time. This topic is important because soil C stocks are large, and soil C flux estimates contribute substantially to total forest C flux, as shown in Table 7-6 and Figure 7-2.

The estimates of C stored in harvested wood products are currently being revised using more detailed wood products production and use data, and more detailed parameters on disposition and decay of products.

N_2O Fluxes from Soils (IPCC Source Category 5.A.1)

Of the fertilizers applied to soils in the United States, no more than one percent is applied to forest soils. Application rates are similar to those occurring on cropped soils, but in any given year, only a small proportion of total forested land receives fertilizer. This is because forests are typically fertilized only twice during their approximately 40 year growth cycle (once at planting and once at approximately 20 years). Thus, although the rate of fertilizer application for the area of forests that receives fertilizer in any given year is relatively high, average annual applications, inferred by dividing all forest land by the amount of fertilizer added to forests in a given year, is quite low. Nitrous oxide (N_2O) emissions from forest soils for 2003 were almost 7 times higher than the baseline year (1990). The trend toward increasing N_2O emissions is a result of an increase in fertilized area of pine plantations in the southeastern United States. Total 2003 forest soil N_2O emissions are roughly equivalent to 3.3 percent of the total forest soil carbon flux, and 0.07 percent of the total sequestration in standing forests, and are summarized in Table 7-9.

Table 7-9. N_2O Fluxes from Soils in Forests Remaining Forests (Tg CO_2 Eq. and Gg)

Forests Remaining Forests: N_2O Fluxes	1990		1997	1998	1999	2000	2001	2002	2003

from Soils									
Tg CO_2 Eq.	0.06		0.30	0.35	0.47	0.35	0.39	0.39	0.39
Gg	0.19		0.96	1.14	1.50	1.14	1.26	1.26	1.26

Methodology

According to U.S. Forest Service statistics for 1996 (USDA Forest Service 2001), approximately 75 percent of trees planted for timber, and about 60 percent of national total harvested forest area are in the southeastern United States. Consequently, it was assumed that southeastern pine plantations represent the vast majority of fertilized forests in the United States. Therefore, estimates of direct N_2O emissions from fertilizer applications to forests were based on the area of pine plantations receiving fertilizer in the southeastern United States and estimated application rates (North Carolina Sate Forest Nutrition Cooperative 2002). Not accounting for fertilizer applied to non-pine plantations is justified because fertilization is routine for pine forests but rare for hardwoods (Binkley et al. 1995). For each year, the area of pine receiving N fertilizer was multiplied by the midpoint of the reported range of N fertilization rates (150 lbs. N per acre). Data for areas of forests receiving fertilizer outside the southeastern United States were not available, so N additions to non-southeastern forests are not included here; however, it should be expected that emissions from the small areas of fertilized forests in other regions would be insubstantial because the majority of trees planted and harvested for timber are in the southeastern United States (USDA Forest Service 2001). Area data for pine plantations receiving fertilizer in the southeast were not available for 2002 and 2003, so data from 2001 were substituted for these years. The proportion of N additions that volatilized from forest soils was assumed to be 10 percent of total amendments, according to the IPCC's default. The unvolatilized N applied to forests was then multiplied by the IPCC default emission factor of 1.25 percent to estimate direct N_2O emissions. The volatilization and leaching/runoff fractions, calculated according to the IPCC default factors of 10 percent and 30 percent, respectively, were included with all sources of indirect emissions in the Agricultural Soil Management source category of the Agriculture sector.

Uncertainty

The amount of N_2O emitted from forests depends not only on N inputs, but also on a large number of variables, including organic carbon availability, O_2 partial pressure, soil moisture content, pH, temperature, and tree planting/harvesting cycles. The effect of the combined interaction of these variables on N_2O flux is complex and highly uncertain. The IPCC default methodology used here does not incorporate any of these variables and only accounts for variations in estimated fertilizer application rates and estimated areas of forested land receiving fertilizer. All forest soils are treated equivalently under this methodology. Furthermore, only synthetic fertilizers are captured, so applications of organic fertilizers are not accounted for here.

Uncertainties exist in the fertilizer application rates, the area of forested land receiving fertilizer, and the emission factors used to derive emission estimates. Uncertainty was calculated according to a modified IPCC Tier 1 methodology. The 95 percent confidence interval of the IPCC default emission factor for synthetic fertilizer applied to soil, according to Chapter 4 of IPCC (2000), ranges from 0.25 to 6 percent. While a Tier 1 analysis should be generated from a symmetrical distribution of uncertainty around the emission factor, an asymmetrical distribution was imposed here to account for the fact that the emission used was not the mean of the range given by IPCC. Therefore, an upper bound of 480 percent and a lower bound of 80 percent were assigned to the emission factor. The higher uncertainty percentage is shown below, but the lower bound reflects a truncated distribution. The uncertainties in the area of forested land receiving fertilizer and fertilization rates were conservatively estimated to be ±54 percent (Binkley 2004). The results of the Tier 1 quantitative uncertainty analysis are summarized in Table 7-10. N_2O fluxes from soils were estimated to be between 0.01 and 2.3 Tg CO_2 Eq. at a 95 percent confidence level. This indicates a range of 96 percent below and 483 percent above the 2003 emission estimate of 0.4 Tg CO_2 Eq.

Table 7-10: Tier 1 Quantitative Uncertainty Estimates of N_2O Fluxes from Forest Soils (Tg CO_2 Eq. and Percent)

IPCC Source Category	Gas	Year 2003 Emissions (Tg CO_2 Eq.)	Uncertainty (%)	Uncertainty Range Relative to 2003 Emission Estimate (Tg CO_2 Eq.)

				Lower Bound	Upper Bound
Forests Remaining Forests: N_2O Fluxes from Soils	N_2O	0.4	96 to 483%	0.01	2.3

Recalculations Discussion

The current Inventory reports N_2O emissions from soils in forested areas separately for the first time. In previous Inventories, N_2O emissions from this source were implicitly included with N_2O emissions from agricultural soils. The net effect of separating forest soils from agricultural soils for the current Inventory is to reduce emissions reported from the agricultural sector by a very small amount. However, because the methods for reporting that source have changed significantly this year, it is impossible to isolate the magnitude of change caused by this recalculation alone on the overall differences in N_2O emissions from agricultural soils. The 2003 direct emission estimate for N_2O from forest soils amounts to an offset of total forest carbon sequestration of approximately 0.5 percent (including standing forests and wood products).

7.2. Land Converted to Forest Land (Source Category 5A2)

Land-use change is constantly occurring, and areas under a number of differing land use types are converted to forest each year, just as forest lands are converted to other uses. However, the magnitude of these changes is not currently known. Given the paucity of available land use information relevant to this particular IPCC source category, it is not possible to quantify CO_2 or N_2O fluxes from land converted to forest land at this time.

7.3. Croplands Remaining Croplands

Changes in Agricultural Soil Carbon Stocks (IPCC Source Category 5B1)

Soils contain both organic and inorganic forms of carbon (C) that contribute to the total soil carbon stock. It is the organic soil carbon (SOC) stocks in mineral and organic soils that may respond to management practices by producing or sequestering greenhouse gases. The IPCC methodology for estimating impacts of agricultural practices on soil C stocks (IPCC/UNEP/OECD/IEA 1997) is divided into three categories of land-use/land-management activities: 1) agricultural land-use and land-management activities on mineral soils; 2) agricultural land-use and land-management activities on organic soils; and 3) liming of soils. Nitrous oxide emissions from agricultural soils are presented within the Agriculture sector.

Mineral soils contain comparatively low amounts of organic C, much of which is concentrated near the soil surface. Typical well-drained mineral surface soils contain from 1 to 6 percent organic C (by weight), although some mineral soils that experience long-term saturation during the year may contain significantly more C (NRCS 1999). Mineral subsoils contain even lower amounts of organic C (NRCS 1999, Brady and Weil 1999). When mineral soils undergo conversion from their native state to agricultural use, as much as half the SOC can be lost to the atmosphere. The rate and ultimate magnitude of C loss will depend on native vegetation, conversion method and subsequent management practices, climate, and soil type. In the tropics, 40 to 60 percent of the C loss generally occurs within the first 10 years following conversion; after that, C stocks continue to decline but at a much slower rate. In temperate regions, C loss can continue for several decades. Eventually, the soil will reach a new equilibrium that reflects a balance between C accumulation from plant biomass and C loss through oxidation. Any changes in land-use or management practices that result in increased organic inputs or decreased oxidation of organic C (e.g., improved crop rotations, cover crops, application of organic amendments and manure, and reduction or elimination of tillage) will result in a net accumulation of SOC until a new equilibrium is achieved.

Organic soils, also referred to as histosols, include all soils with more than 12 to 20 percent organic C by weight, depending on clay content (NRCS 1999, Brady and Weil 1999). The organic layer of these soils is also typically extremely deep. Organic soils form under waterlogged conditions, in which minimal decomposition of plant residue occurs. When organic soils are cultivated, they are first drained which, together with tilling or mixing of the soil, aerates the soil, and thereby accelerates the rate of decomposition and CO_2 generation. Because of the depth and richness of the organic layers, C loss from cultivated organic soils can continue over long periods of time.

When organic soils are disturbed for cultivation purposes, which invariably include drainage, the rate at which organic matter decomposes and CO_2 emissions are generated, is determined primarily by climate, composition (i.e., decomposability) of the organic matter, and the specific land-use practices undertaken. The use of organic soils for annual crop production results in greater C loss than conversion to pasture or forests, due to deeper drainage and more intensive management practices (Armentano and Verhoeven 1990, as cited in IPCC/UNEP/OECD/IEA 1997).

The last category of the IPCC methodology addresses emissions from lime (in the form of crushed limestone ($CaCO_3$)) and dolomite ($CaMg(CO_3)_2$) additions to agricultural soils. Lime and dolomite are added by land managers to ameliorate acidification. When these compounds come in contact with acid soils, they degrade, thereby generating CO_2. Complete degradation of applied limestone and dolomite could take several years, but it could also take significantly less time, depending on the soil conditions and the type of mineral applied.

The estimates in this section include management impacts on mineral and organic soil C stocks for croplands and grasslands. Due to limited data availability, it is impossible to differentiate between the stock changes resulting from management practices on croplands or grasslands and those resulting from conversions between the two (i.e., statistics have been developed in a manner that only net changes in conversions are determined with no tracking of the total amount of land converted between cropland and grassland uses).

Total SOC stock depends on the balance between inputs of organic material (e.g., decayed plant matter, roots, and organic amendments such as manure and crop residues) and loss of C through decomposition. The quantity and quality of organic matter inputs and their rate of decomposition are determined by the combined interaction of climate, soil properties, and land use. Agricultural practices such as clearing, drainage, tillage, planting, grazing, crop residue management, fertilization, and flooding, can modify both organic matter inputs and decomposition, and thereby result in a net flux of C to or from soils.

Of the three activities (those associated with mineral soils, organic soils, and liming of soils) land use and management of mineral soils was the most important component of total flux during the 1990 through 2003 period. C sequestration in mineral soils in 2003 was estimated to be approximately 51.7 Tg CO_2 Eq. (14 Tg C), while emissions from organic soils were estimated to be 35.6 Tg CO_2 Eq. (10 Tg C), and emissions from the practice of liming were estimated at 9.5 Tg CO_2 Eq. (3 Tg C). Together, the three activities accounted for net sequestration of approximately 6.6 Tg CO_2 Eq. (2 Tg C) in 2003. Total annual net CO_2 flux was negative (i.e., net sequestration occurred) each year over the inventory period, although the net C storage in soils did decline by 18 percent between 1990 and 2003. Net sequestration was largely due to annual cropland enrolled in the Conservation Reserve Program, cropland converted to permanent pastures and hay production, a reduction in the frequency of summer-fallow use in semi-arid areas, and some increase in the adoption of conservation tillage (i.e., reduced and no till practices). The decline in net sequestration was attributed to two management practices. First, the amount of organic soils that were drained for agricultural production increased during the time period, thus leading to higher emissions. Second, manure production declined over the inventory period, particularly during the last few years, and this decline reduced the amount of crop and grazing lands receiving organic amendments, and thus there was a small decline in C sequestration attributed to this activity.

The spatial variability in annual CO_2 flux for mineral and organic soils is displayed in Figure 7-4 through Figure 7-7. The highest rates of sequestration occur mostly in the southern and northern Great Plains, southern portions of the corn-belt in the Midwest, and the lower Mississippi River Valley. Sequestration rates are also relatively high in the southeastern United States. These regions either have high Conservation Reserve Program enrollment (particularly the Great Plains region), and/or have adopted conservation tillage at a higher rate than other regions of the country. The greatest organic soil emission rates are from drained peatlands along the southeastern coastal region, in the northeast central United States surrounding the Great Lakes, and along the central and northern portions of the west coast.

Table 7-11: Net CO_2 Flux from Agricultural Soils (Tg CO_2 Eq.)

Soil Type	1990		1997	1998	1999	2000	2001	2002	2003
Mineral Soils	(52.4)		(51.7)	(49.5)	(48.9)	(50.0)	(51.6)	(51.9)	(51.7)
Organic Soils	34.8		35.6	35.6	35.6	35.6	35.6	35.6	35.6
Liming of Soils	9.5		8.7	9.6	9.1	8.8	9.0	10.1	9.5

Total Net Flux	**(8.1)**		**(7.4)**	**(4.3)**	**(4.3)**	**(5.7)**	**(7.1)**	**(6.2)**	**(6.6)**

Note: Parentheses indicate net sequestration. Shaded areas indicate values based on a combination of historical data and projections. All other values are based on historical data only. Estimates include the change in C storage resulting from the annual application of sewage sludge and the change in manure amendments or Conservation Reserve Program enrollment after 1997.

Table 7-12: Net Carbon Flux from Agricultural Soils (Tg C)

Soil Type	**1990**		**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**
Mineral Soils	(14.3)		(14.1)	(13.5)	(13.3)	(13.6)	(14.1)	(14.1)	(14.1)
Organic Soils	9.5		9.7	9.7	9.7	9.7	9.7	9.7	9.7
Liming of Soils	2.6		2.4	2.6	2.5	2.4	2.4	2.8	2.6
Total Net Flux	**(2.2)**		**(2.0)**	**(1.2)**	**(1.2)**	**(1.5)**	**(1.9)**	**(1.7)**	**(1.8)**

Note: Parentheses indicate net sequestration. Shaded areas indicate values based on a combination of historical data and projections. All other values are based on historical data only. Estimates include the change in C storage resulting from the annual application of sewage sludge and the change in manure amendments or Conservation Reserve Program enrollment after 1997.

Figure 7-4: Net Annual CO_2 Flux, per Hectare, From Mineral Soils Under Agricultural Management, 1990-1992

Figure 7-5: Net Annual CO_2 Flux, per Hectare, From Mineral Soils Under Agricultural Management, 1993-2003

Figure 7-6: Net Annual CO_2 Flux, per Hectare, From Organic Soils Under Agricultural Management, 1990-1992

Figure 7-7: Net Annual CO_2 Flux, per Hectare, From Organic Soils Under Agricultural Management, 1993-2003

The flux estimates presented here are restricted to CO_2 fluxes associated with the use and management of agricultural soils. Agricultural soils are also important sources of other greenhouse gases, particularly N_2O from application of fertilizers, manure, and crop residues and from cultivation of legumes, as well as methane (CH_4) from flooded rice cultivation. These emissions are accounted for under the Agriculture sector.[5]

Methodology

The methodologies used to calculate net CO_2 flux from use and management of mineral and organic soils and from liming follow the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997, Ogle et al. 2002, Ogle et al. 2003), except where noted below. Additional details on the methodology and data used to estimate flux from mineral and organic soils are described in Annex 3.13.

Mineral and Organic Soils

Mineral SOC stocks were estimated for 1982, 1992, and 1997, as well as CO_2 emissions from organic soils, for the conterminous United States and Hawaii using U.S. data on climate, soil types, land use and land management activity data, reference C stocks (for agricultural soils rather than native soils) and field studies addressing management effects on SOC storage. National-scale data on land-use and management changes over time were obtained from the *1997 National Resources Inventory* (USDA-NRCS 2000). The *1997 National Resources Inventory* provides land use/management data and soils information for more than 400,000 locations in U.S. agricultural lands. Two other sources were used to supplement the land-use information from the *1997 National*

[5] Nitrous oxide emissions from agricultural soils and methane emissions from rice fields are addressed under the Agricultural Soil Management and Rice Cultivation sections, respectively, of the Agriculture sector.

Resources Inventory. The Conservation Technology Information Center (CTIC 1998) provided data on tillage activity, with adjustments for long-term adoption of no-till agriculture (Towery 2001), and Euliss and Gleason (2002) provided activity data on wetland restoration of Conservation Reserve Program Lands. Manure N production was derived from USDA livestock population data (USDA 1994a,b; 1995a,b; 1998a,b; 1999a-e; 2000a-g; 2001a-f; 2002a-f; 2003a-f), the FAOSTAT database (FAO 2003), and Lange (2000). Manure management information was obtained from Poe et al. (1999), Safley et al. (1992), and personal communications with agricultural experts (Anderson 2000, Deal 2000, Johnson 2000, Miller 2000, Milton 2000, Stettler 2000, Sweeten 2000, Wright 2000). Livestock weight data were obtained from Safley (2000), USDA (1996, 1998c), and ASAE (1999); daily rates of N excretion from ASAE (1999) and USDA (1996); and information about the fraction of poultry litter used as a feed supplement from Carpenter (1992).

For estimating the emissions from both mineral and organic soils, Major Land Resource Areas were used as the base spatial unit for mapping climate regions in the United States. Each Major Land Resource Area represents a geographic unit with relatively similar soils, climate, water resources, and land uses (NRCS 1981).[6] Major Land Resource Areas were classified into climate zones according to the IPCC categories using the Parameter-Evaluation Regressions on Independent Slopes Model (PRISM) climate-mapping program of Daly et al. (1994).

For mineral soils, reference C stocks were estimated using the National Soil Survey Characterization Database (NRCS 1997) with cultivated cropland as the reference condition, rather than native vegetation as used in the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). Changing the reference condition was necessary because soil measurements under agricultural management are much more common and easily identified in the National Soil Survey Characterization Database (NRCS 1997) than those which are not considered cultivated cropland. U.S. management factors[7] were derived from published literature to determine the impact of management practices on SOC storage, including changes in tillage, cropping rotations and intensification, land-use change between cultivated and uncultivated conditions, as well as C loss rates associated with drainage of organic soils under agricultural management (Ogle et al. 2003, Ogle et al. in review). U.S. management factors associated with organic matter amendments and improving grazing lands were not estimated because of few studies analyzing those impacts. Instead, IPCC factors from *LULUCF Good Practice Guidance* (IPCC 2003) formed the basis for quantifying the effect of those activities. Euliss and Gleason (2002) provided the data for computing the change in SOC storage resulting from restoration of Conservation Reserve Program lands (Olness et al. in press, Euliss et al. in prep).

Combining information from these data sources, SOC stocks for mineral soils were estimated 50,000 times for 1982, 1992, and 1997, using a Monte Carlo simulation approach and the probability density functions for U.S.-specific management factors, reference C stocks, and land-use activity data (Ogle et al. 2003, Ogle et al. 2002). The annual C flux for 1990 through 1992 was determined by calculating the annual change in stocks between 1982 and 1992; annual C flux for 1993 through 2003 was determined by calculating the annual change in stocks between 1992 and 1997. Annual C flux estimates for mineral soils between 1990 and 2003 were adjusted to account for additional C sequestration from sewage sludge applications, as well as gains or losses in C sequestration after 1997 due to changes in Conservation Reserve Program enrollment and manure N production. For the entire inventory period, the amount of land amended with sewage sludge was estimated from N application data from the Soil Management section of the Agriculture chapter of this volume, and an assumed application rate derived from Kellogg et al. (2000). To estimate the impact of manure amendments after 1997, the change in manure N production was determined relative to the amount produced in 1997, and then similar to sewage sludge calculations, the production values were multiplied by the assumed application rate to determine the change in land area that was amended with manure. Carbon storage rate was estimated at 0.22 metric tons C per hectare per year for both the manure and sewage sludge amendments. To estimate the impact of enrollment in the Conservation Reserve

[6] The polygons displayed in Figure 6-5 through Figure 6-8 are the Major Land Resource Areas.

[7] Management factors have been derived from published literature to reflect changes in tillage, cropping rotations and intensification, land-use change between cultivated and uncultivated conditions, as well as drainage of organic soils.

Program after 1997, the change in enrollment acreage relative to 1997 was derived based on Barbarika (2004), and the differences in mineral soil areas were multiplied by 0.5 metric tons C per hectare per year.

Annual C emission estimates from organic soils between 1990 and 2002 were derived using *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997), except that U.S.-specific C loss rates were used in the calculations rather than default IPCC rates (Ogle et al. 2003). Similar to mineral soils, the final estimates included a measure of uncertainty as determined from the Monte Carlo simulation with 50,000 iterations. Emissions were based on the 1992 and 1997 land areas from the *1997 National Resources Inventory* (USDA-NRCS 2000). The annual flux estimated for 1992 was applied to 1990 through 1992, and the annual flux estimated for 1997 was applied to 1993 through 2003.

Liming

Carbon dioxide emissions from degradation of limestone and dolomite applied to agricultural soils were calculated by multiplying the annual amounts of limestone and dolomite applied (see Table 7-13) by CO_2 emission factors (0.120 metric ton C/metric ton limestone, 0.130 metric ton C/metric ton dolomite) (IPCC 2003).[8] These emission factors are based on the assumption that all of the C in these materials evolves as CO_2 in the same year in which the minerals are applied. The annual application rates of limestone and dolomite were derived from estimates and industry statistics provided in the *Minerals Yearbook* and *Mineral Industry Surveys* (Tepordei 1993, 1994, 1995, 1996, 1997, 1998, 1999, 2000, 2001, 2002, 2003, 2004; USGS 2002, 2003, 2004). To develop these data, USGS (U.S. Bureau of Mines prior to 1997) obtained production and use information by surveying crushed stone manufacturers. Because some manufacturers were reluctant to provide information, the estimates of total crushed limestone and dolomite production and use were divided into three components: 1) production by end-use, as reported by manufacturers (i.e., "specified" production); 2) production reported by manufacturers without end-uses specified (i.e., "unspecified" production); and 3) estimated additional production by manufacturers who did not respond to the survey (i.e., "estimated" production).

To estimate the "unspecified" and "estimated" amounts of crushed limestone and dolomite applied to agricultural soils, the fractions of "unspecified" and "estimated" production that were applied to agricultural soils in a specific year were assumed to be equal to the fraction of "specified" production that was applied to agricultural soils in that same year. In addition, data were not available for 1990, 1992, and 2003 on the fractions of total crushed stone production that were limestone and dolomite, and on the fractions of limestone and dolomite production that were applied to soils. To estimate the 1990 and 1992 data, a set of average fractions were calculated using the 1991 and 1993 data. These average fractions were applied to the quantity of "total crushed stone produced or used" reported for 1990 and 1992 in the 1994 *Minerals Yearbook* (Tepordei 1996). To estimate 2003 data, the previous year's fractions were applied to a 2003 estimate of total crushed stone presented in the USGS *Mineral Industry Surveys: Crushed Stone and Sand and Gravel in the First Quarter of 2004* (USGS 2004).

The primary source for limestone and dolomite activity data is the *Minerals Yearbook*, published by the Bureau of Mines through 1994 and by the U.S. Geological Survey from 1995 to the present. In 1994, the "Crushed Stone" chapter in *Minerals Yearbook* began rounding (to the nearest thousand) quantities for total crushed stone produced or used. It then reported revised (rounded) quantities for each of the years from 1990 to 1993. In order to minimize the inconsistencies in the activity data, these revised production numbers have been used in all of the subsequent calculations.

Table 7-13: Quantities of Applied Minerals (Thousand Metric Tons)

Mineral	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003

[8] The default emission factor for dolomite provided in the Workbook volume of the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997) and the *Good Practice Guidance for Land Use, Land-Use Change, and Forestry* (IPCC 2003) is incorrect. The value provided is 0.122 metric ton carbon/metric ton of dolomite; the correct value is 0.130 metric ton carbon/metric ton of dolomite.

Limestone	19,012	20,312	17,984	15,609	16,686	17,297	17,479	16,539	14,882	16,894	15,863	16,097	20,449	19,163
Dolomite	2,360	2,618	2,232	1,740	2,264	2,769	2,499	2,989	6,389	3,420	3,812	3,951	2,353	2,205

Uncertainty

Uncertainties for mineral and organic soils were quantified using a Monte Carlo Approach by constructing probability distribution functions (PDF) for inputs to the IPCC equations, including management factors, C emission rates for organic soils, and land use and management activity data, and then simulating a range of values using the Monte Carlo framework (Ogle et al. 2003, Annex 3.13). Uncertainty estimates do not include sewage sludge impacts on SOC storage for any year in the inventory period, or contributions of changing manure management and enrollment in the Conservation Reserve Program after 1997. PDFs for management factors were derived from a synthesis of 91 published studies, which addressed the impact of management on SOC storage. Uncertainties in land use and management activity data were also derived from a statistical analysis. The National Resources Inventory (NRI) has a two-stage sampling design that allowed PDFs to be constructed assuming a multivariate normal distribution accounting for dependencies in activity data. PDFs for the tillage activity data, as provided by the Conservation Technology and Information Center, were constructed on a bivariate normal distribution with a log-ratio scale, accounting for the negative dependence among the proportions of land under conventional and conservation tillage practices. PDFs for the crop and grazing land area receiving manure amendments were based on the data sources given for manure amendments in the methodology section, and a statistical relationship between production and the amount of land area that is amended according to manure management information derived from the USDA Census of Agriculture (Edmonds et al. 2003). Lastly, enrollment in wetland restoration programs was estimated from contract agreements, but due to a lack of information, PDFs were constructed assuming a nominal ±50 percent uncertainty range.

The results of the Tier 2 quantitative uncertainty analysis are summarized in Table 7-14. CO_2 flux from mineral and organic agricultural soil carbon stocks in 2003 was estimated to be between -40.0 and +5.9 Tg CO_2 Eq. at a 95 percent confidence level (or 19 of 20 Monte Carlo Stochastic Simulations). This indicates a range of 148 percent below to 136 percent above the 2003 flux estimate of -16.1 Tg CO_2 Eq.

Table 7-14: Tier 2 Quantitative Uncertainty Estimates for CO_2 Flux from Mineral and Organic Agricultural Soil Carbon Stocks (Tg CO_2 Eq. and Percent)

			Uncertainty Range Relative to Flux Estimate[a]			
			(Tg CO_2 Eq.)		(%)	
Source	Gas	2003 Flux Estimate (Tg CO_2 Eq.)	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Mineral and Organic Soil Uncertainty	CO_2	(16.1)	(40.0)	5.9	-148%	+136%

[a] Includes mineral and organic soils only; estimates do not include the change in C storage resulting from the annual application of sewage sludge, or the change in manure amendments or Conservation Reserve Program enrollment after 1997.
[b] Range of emission estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval.

The time-series calculations were consistent for each reporting year of the inventory in terms of methodology, with the only difference in reported values stemming from the changes in land use and management activities across U.S. agricultural lands. In addition, the same management factors (i.e., emission factors) were used each year for calculating the impact of land use and management on SOC stocks. There is no evidence that changing management practices has a quantitatively different impact on SOC stocks over the inventory period. For example, changing from conventional to no-till management in 1990 is assumed to have the same impact on soil C stocks over the course of this first year as it is over the course of each year in the 20 year period following the management change.

Although the mineral and organic soil estimates have been improved during the last two years using a Monte Carlo approach with the incorporation of U.S.-specific reference C stocks management factor values, and a more comprehensive accounting of manure amendment impacts on SOC storage, several limitations remain in the analysis. First, minimal data exist on where and how much sewage sludge has been applied to U.S. agricultural

lands and the accounting of this activity appears to be much more difficult than the related-activity of using manure to amend agricultural soils. Consequently, uncertainties have not been estimated for the change in SOC storage resulting from sludge applications. Second, due to the IPCC requirement that inventories include all land areas that are potentially subject to land-use change, the *1997 National Resources Inventory* dataset includes some points designated as non-agricultural land-uses if the points were once categorized as agricultural land use and their designation changed during the period from 1992 to 1997. The non-agricultural land uses are urban, water, and miscellaneous non-cropland (e.g., roads and barren areas). The impact on SOC storage that results from converting cropland to non-agricultural uses is not well-understood, and therefore, those points were not included in the calculations for mineral soils (emissions from organic soils, however, were computed for those points in the years that they were designated as an agricultural use). Third, the current estimates may underestimate losses of C from organic soils because the *1997 National Resources Inventory* was not designed as a soil survey and organic soils frequently occur as relatively small inclusions within major soil types. Lastly, this methodology does not take into account changes in SOC stocks due to pre-1982 land use and land-use change.

Uncertainties in the estimates of emissions from liming result from both the methodology and the activity data. The IPCC method assumes that all inorganic C in the applied minerals evolves to CO_2, and that this degradation occurs in the same year that the minerals are applied. However, recent research has shown that liming can either be a C source or a sink, depending upon weathering reactions, which are pH dependent (Hamilton et al. 2002). Moreover, it can take several years for agriculturally applied limestone and dolomite to degrade completely. However, application rates are fairly constant over the entire time series, so this latter assumption may not contribute significantly to overall uncertainty.

There are several sources of uncertainty in the limestone and dolomite activity data. When reporting data to the USGS (or U.S. Bureau of Mines), some producers do not distinguish between limestone and dolomite. In these cases, data are reported as limestone, so this reporting could lead to an overestimation of limestone and an underestimation of dolomite. In addition, the total quantity of crushed stone listed each year in the *Minerals Yearbook* excludes American Samoa, Guam, Puerto Rico, and the U.S. Virgin Islands.

Uncertainty regarding lime applications was estimated at 15 percent (Tepordei 2003). While IPCC provides no uncertainty values for the emission factor from this subsource, there is uncertainty stemming primarily from the inherent assumption that all applications are converted to CO_2 within the year of application. In fact, limestone may persist in the soil for 3 to 4 years following application (Nardozzi 2004), indicating that emissions may continue throughout that period of time. Taking this into account, the resulting uncertainty in the emission factor is estimated to be 75 percent. The preliminary results of the Tier 1 quantitative uncertainty analysis are summarized in Table 7-15. Liming of agricultural soils CO_2 emissions in 2003 were estimated to be between 2.2 and 10.9 Tg CO_2 Eq. at the 95 percent confidence level. This indicates a range of 76 percent above and below the 2003 emission estimate of 9.5 Tg CO_2 Eq.

Table 7-15: Tier 1 Quantitative Uncertainty Estimates for CO_2 Emissions from Liming of Agricultural Soils (Tg CO_2 Eq. and Percent)

IPCC Source Category	Gas	Year 2003 Emissions (Tg CO_2 Eq.)	Uncertainty (%)	Uncertainty Range Relative to 2003 Emission Estimate (Tg CO_2 Eq.)	
				Lower Bound	Upper Bound*
Liming of Agricultural Soils	CO_2	9.5	76%	2.2	10.9

* Because the current Inventory methodology assumes that all of the limestone and dolomite decomposes in the year of application, the emission factors could be significantly lower, but could not be higher. Consequently, the emissions estimate may only be higher due to the uncertainty in the application data. Therefore, while normally, Tier 1 analyses generate a symmetrical distribution of uncertainty around the emission estimate, an asymmetrical distribution was necessarily imposed here.

It is not currently possible to combine the results of this Tier 1 uncertainty analysis with those of the Tier 2 uncertainty analyses for CO_2 fluxes from mineral soils and histosols.

Recalculations Discussion

The estimates of changes in agricultural SOC stocks have been modified in several ways. First, uncertainty in manure amendments was evaluated and incorporated into the general inventory calculations for agricultural soil C, using the Tier 2 IPCC methodology (IPCC/UNEP/OECD/IEA 1997). In previous Inventories, the change in SOC storage attributed to organic amendments was estimated as a post-analysis calculation using a simplistic activity-based approach. Incorporating amendment data into the IPCC analysis led to a decline in the estimated change for SOC storage attributed to manure management, relative to previous years. The activity-based approach assumed that all of the applied manure was leading to an increase in SOC, while the IPCC method only accounted for the changes in organic amendments from the baseline year to estimate the impact on SOC storage. In general, the IPCC approach assumes that impacts on SOC storage are manifested in the first 20 years following a management change, and consequently, this method does not account for smaller residual changes in SOC storage that can occur in later years. However, the activity-based approach also has limitations because it does not account for the possibility that a portion of the manure applied in a year is simply maintaining SOC storage from past applications. In fact, the production of manure has not changed to a large extent since 1982 according to USDA statistics (see Annex 3.13), and, therefore, many of the current amendments are only serving the purpose of maintaining past storage. Hence, incorporating manure management into the IPCC calculations has produced a more conservative estimate by reducing the potential for over-estimating the impact of longer-term manure amendments, but also does not capture the residual change in SOC storage occurring after the first 20 years.

Two additional revisions have been incorporated into the analysis since the previous Inventory. As part of the revision, management factors provided in the IPCC *LULUCF Good Practice Guidance* (IPCC 2003) now form the basis to estimate management impacts for which U.S.-specific factors have not been derived (i.e., improving grazing lands and organic amendments). Previously, defaults factor values from the *Revised 1996 IPCC Guidelines* formed the basis for these calculations using the IPCC method, but greater disaggregation of management factors by climate in the IPCC *LULUCF Good Practice Guidance* was assumed to produce more realistic estimates of those management impacts, compared to the single global values provided in the *Revised 1996 IPCC Guidelines*.

Lastly, emissions from organic soils have changed slightly from those reported in the previous inventory, as a result of revising the land area included in the emission calculation. In previous years, lands that were converted between agricultural and non-agricultural uses (i.e., urban, water, and miscellaneous non-cropland) were not included in the estimation of emissions from organic soils, and this led to a slight under-estimation of emissions from organic soils. Specifically, a small amount of land area under agricultural management in 1992 or 1997 was not included in the emission calculation because it had been converted from a non-agricultural use.

Estimates of CO_2 emissions from agricultural soil management have been revised due to methodological and historical data changes in the calculations of N from livestock that is applied to soils. These changes include corrections to: the typical animal mass value for beef cows and calves; the accounting of sheep in New England states; state broiler populations; and updated NASS animal population estimates for the years 1998 through 2001 (NASS 2000). Additionally, the factor for converting short tons to metric tons was revised to include another significant digit, and the percent residue applied for rice in the year 2001 was corrected. In combination, these changes resulted in a minor effect on the agricultural soil C estimates with a reduction in the CO_2 sink by less than 1 percent.

The quantity of applied minerals reported in the previous inventory for 2002 has been revised. Consequently, the reported emissions resulting from liming in 2002 have also changed. In the previous inventory, to estimate 2002 data, the previous year's fractions were applied to a 2002 estimate of total crushed stone presented in the USGS *Mineral Industry Surveys: Crushed Stone and Sand and Gravel in the First Quarter of 2003* (USGS 2003). Since publication of the previous inventory, the *Minerals Yearbook* has published actual quantities of crushed stone sold or used by producers in the United States in 2002. These values have replaced those used in the previous inventory to calculate the quantity of minerals applied to soil and the emissions from liming.

Overall, these changes resulted in an average annual decrease of 14.2 Tg CO_2 Eq. (67 percent) in agricultural soil carbon stocks for the period 1990 through 2002.

Planned Improvements

A major planned improvement is currently underway that will enhance reporting of changes in agricultural soil C stocks and deals with an alternative inventory approach to better represent between-year variability in annual fluxes. This new annual activity-based inventory will use the Century ecosystem simulation model, which relies on actual climate, soil, and land use/management databases to estimate variation in fluxes. This inventory will provide a more robust accounting of C stock changes in U.S. agricultural lands than the more simplistic IPCC soil C accounting approach. This approach is likely to be used in the future for reporting of land use and management impacts on agricultural soil C stocks, and therefore a short description of this method compared to the IPCC approach is provided.

The Century ecosystem model has been widely tested and found to be successful in simulating those processes affecting SOC storage (Metherell et al. 1993, Parton et al. 1994). Simulation modeling differs from the IPCC approach in that annual changes are computed dynamically as a function of inputs of C and N to soil (e.g., crop residues, manure) and C emissions from organic matter decomposition, which are governed by climate and soil factors as well as management practices. The model distinguishes between all major field crops (maize, wheat and other small grains, soybean, sorghum, cotton) as well as hay and pasture (grass, alfalfa, clover). Management variables include tillage, fertilization, irrigation, drainage, and manure addition.

Input data are largely derived from the same sources as the IPCC-based method (i.e., climate variables come from the Parameter-Evaluation Regressions on Independent Slopes Models (PRISM) database; crop rotation, irrigation and soil characteristics from the National Resources Inventory (NRI); and tillage data from the Conservation Technology Information Center (CTIC)). In addition, the Century analysis uses detailed information on crop rotation-specific fertilization and tillage implements obtained from USDA's Economic Research Service. The main difference between the methods is that the climate, soil, and management data serve as driving variables in the Century simulation, whereas in the IPCC approach these data are more highly aggregated and are used for classification purposes. In the Century-based analysis, land areas having less than 5 percent of total area in crop production are excluded and several less-dominant crops (e.g., vegetables, sugar beets and sugar cane, potatoes, tobacco, orchards, and vineyards), for which the model has not yet been parameterized, are not included. Thus, the total area included in the Century analysis (149 million hectares) will be smaller than the corresponding area of cropland (165 million hectares) included in the IPCC estimates.

Preliminary results using the Century model suggest (as with the IPCC model) that U.S. cropland mineral soils (excluding organic soils) are currently acting as a C sink. The Century model estimates that U.S. cropland soils sequestered an average of approximately 77 Tg CO_2 Eq. annually (21 Tg C/year) for 1992 through 1997. Organic soils (which contribute large C losses) have not yet been simulated by Century.

As with the IPCC method, increases in mineral SOC stocks in the Century analysis are associated with reduced tillage, Conservation Reserve Program lands, reduced bare fallow and some increase in hay area. However, the Century analysis also includes the effect of a long-term trend in increasing residue inputs due to higher productivity on cropland in general, contributing to increasing SOC stocks. Work is underway to refine model input data and to estimate uncertainty for the dynamic model approach.

Potential advantages of a dynamic simulation-based approach include the ability to use actual observed weather, observed annual crop yields, and more detailed soils and management information to drive the estimates of soil C change. This would facilitate annual estimates of SOC stock changes and CO_2 emissions from soils that would better reflect interannual variability in cropland production and weather influences on C cycle processes.

7.4. Lands Converted to Croplands (Source Category 5B2)

Land-use change is constantly occurring, and areas under a number of differing land use types are converted to croplands each year, just as croplands are converted to other uses. However, while the C stocks for Land Converted to Cropland are included in the Croplands Remaining Croplands section, it is not possible to sub-divide it and determine the magnitude of the change at this time. Additionally, given the lack of available land use information

relevant to this particular IPCC source category, it is not possible to quantify the N_2O flux from Lands Converted to Cropland at this time.

7.5. Settlements Remaining Settlements

Changes in Yard Trimming and Food Scrap Carbon Stocks in Landfills (IPCC Source Category 5E1)

As is the case with carbon in landfilled forest products, carbon contained in landfilled yard trimmings and food scraps can be stored for very long periods. In the United States, yard trimmings (i.e., grass clippings, leaves, and branches) and food scraps comprise a significant portion of the municipal waste stream, and a large fraction of the collected yard trimmings and food scraps are discarded in landfills. However, both the amount of yard trimmings and food scraps collected annually and the fraction that is landfilled have declined over the last decade. In 1990, nearly 51 million metric tons (wet weight) of yard trimmings and food scraps were generated (i.e., put at the curb for collection or taken to disposal or composting facilities) (EPA 2003). Since then, programs banning or discouraging disposal have led to an increase in backyard composting and the use of mulching mowers, and a consequent 20 percent decrease in the amount of yard trimmings collected. At the same time, a dramatic increase in the number of municipal composting facilities has reduced the proportion of collected yard trimmings that are discarded in landfills—from 72 percent in 1990 to 34 percent in 2003. There is considerably less centralized composting of food scraps; generation has grown by 26 percent since 1990, though the proportion of food scraps discarded in landfills has decreased slightly from 81 percent in 1990 to 77 percent in 2003. Overall, there has been a decrease in the yard trimmings and food scrap landfill disposal rate, which has resulted in a decrease in the rate of landfill carbon storage to 10.1 Tg CO_2 Eq. in 2003 from 26.0 Tg CO_2 Eq. in 1990 (Table 7-16 and Table 7-17).

Table 7-16: Net Changes in Yard Trimming and Food Scrap Stocks in Landfills (Tg CO_2 Eq.)

Carbon Pool	1990		1996	1997	1998	1999	2000	2001	2002	2003
Yard Trimmings	**(23.2)**		**(11.3)**	**(10.4)**	**(9.6)**	**(8.5)**	**(7.2)**	**(7.4)**	**(7.5)**	**(7.5)**
Grass	(2.5)		(1.0)	(0.9)	(0.8)	(0.7)	(0.6)	(0.7)	(0.7)	(0.7)
Leaves	(11.2)		(5.9)	(5.4)	(5.1)	(4.5)	(4.0)	(4.0)	(4.0)	(4.0)
Branches	(9.6)		(4.4)	(4.0)	(3.7)	(3.2)	(2.6)	(2.7)	(2.7)	(2.8)
Food Scraps	**(2.8)**		**(2.2)**	**(2.6)**	**(2.9)**	**(2.9)**	**(3.0)**	**(2.9)**	**(2.7)**	**(2.6)**
Total Net Flux	**(26.0)**		**(13.5)**	**(12.9)**	**(12.5)**	**(11.4)**	**(10.2)**	**(10.3)**	**(10.2)**	**(10.1)**

Note: Totals may not sum due to independent rounding.

Table 7-17: Net Changes in Yard Trimming and Food Scrap Stocks in Landfills (Tg C)

Carbon Pool	1990		1996	1997	1998	1999	2000	2001	2002	2003
Yard Trimmings	**(6.3)**		**(3.1)**	**(2.8)**	**(2.6)**	**(2.3)**	**(2.0)**	**(2.0)**	**(2.0)**	**(2.0)**
Grass	(0.7)		(0.3)	(0.2)	(0.2)	(0.2)	(0.2)	(0.2)	(0.2)	(0.2)
Leaves	(3.0)		(1.6)	(1.5)	(1.4)	(1.2)	(1.1)	(1.1)	(1.1)	(1.1)
Branches	(2.6)		(1.2)	(1.1)	(1.0)	(0.9)	(0.7)	(0.7)	(0.7)	(0.8)
Food Scraps	**(0.8)**		**(0.6)**	**(0.7)**	**(0.8)**	**(0.8)**	**(0.8)**	**(0.8)**	**(0.7)**	**(0.7)**
Total Net Flux	**(7.1)**		**(3.7)**	**(3.5)**	**(3.4)**	**(3.1)**	**(2.8)**	**(2.8)**	**(2.8)**	**(2.7)**

Note: Totals may not sum due to independent rounding.

Methodology

Estimates of net carbon flux resulting from landfilled yard trimmings and food scraps were developed by estimating the change in landfilled carbon stocks between inventory years. Carbon stock estimates were calculated by determining the mass of landfilled carbon resulting from yard trimmings or food scraps discarded in a given year; adding the accumulated landfilled carbon from previous years; and subtracting the portion of carbon landfilled in previous years that decomposed.

To determine the total landfilled carbon stocks for a given year, the following were estimated: 1) the composition of the yard trimmings, 2) the mass of yard trimmings and food scraps discarded in landfills, 3) the carbon storage factor of the landfilled yard trimmings and food scraps, and 4) the rate of decomposition of the degradable carbon. The composition of yard trimmings was assumed to be 30 percent grass clippings, 40 percent leaves, and 30 percent branches on a wet weight basis (Oshins and Block 2000). The yard trimmings were subdivided because each component has its own unique carbon storage factor and rate of decomposition. The mass of yard trimmings and food scraps disposed of in landfills was estimated by multiplying the quantity of yard trimmings and food scraps discarded by the proportion of discards managed in landfills. Data on discards (i.e., the amount generated minus the amount diverted to centralized composting facilities) for both yard trimmings and food scraps were taken primarily from *Municipal Solid Waste in the United States: 2001 Facts and Figures* (EPA 2003). That report provides data for 1960, 1970, 1980, 1990, 1995, and 1999 through 2001. To provide data for some of the missing years in the 1990 through 1999 period, two earlier reports were used (*Characterization of Municipal Solid Waste in the United States: 1998 Update* (EPA 1999), and *Municipal Solid Waste in the United States: 2000 Facts and Figures* (EPA 2002)). Remaining years in the time series for which data were not provided were estimated using linear interpolation, except for 2002 and 2003, which was assumed to have the same discards as 2001. These reports do not subdivide discards of individual materials into volumes landfilled and combusted, although they provide an estimate of the proportion of overall wastestream discards managed in landfills and combustors (i.e., ranging from 81 percent and 19 percent respectively in 1990, to 79 percent and 21 percent in 2001).

The amount of carbon disposed of in landfills each year, starting in 1960, was estimated by converting the discarded landfilled yard trimmings and food scraps from a wet weight to a dry weight basis, and then multiplying by the initial (i.e., pre-decomposition) carbon content (as a fraction of dry weight). The dry weight of landfilled material was calculated using dry weight to wet weight ratios (Tchobanoglous et al. 1993 cited by Barlaz 1998) and the initial carbon contents were determined by Barlaz (1998) (Table 7-18).

The amount of carbon remaining in the landfill for each subsequent year was tracked based on a simple model of carbon fate. According to Barlaz (1998), a portion of the initial carbon resists decomposition and is essentially persistent in the landfill environment; the modeling approach applied here builds on his findings. Barlaz (1998) conducted a series of experiments designed to measure biodegradation of yard trimmings, food scraps, and other materials, in conditions designed to promote decomposition (i.e., by providing ample moisture and nutrients). After measuring the initial carbon content, the materials were placed in sealed containers along with a "seed" containing methanogenic microbes from a landfill. Once decomposition was complete, the yard trimmings and food scraps were re-analyzed for carbon content. The mass of carbon remaining, divided by the original dry weight of the material, was reported as the carbon storage factor (Table 7-18).

For purposes of simulating U.S. landfill carbon flows, the carbon storage factors are divided by the initial carbon content to determine the proportion of initial carbon that does not decompose. The remaining portion is assumed to degrade (and results in emissions of CH_4 and CO_2). For example, for branches Barlaz (1998) reported the carbon storage factor as 38 percent (of dry weight), and the initial carbon content as 49 percent (of dry weight). Thus, the proportion of initial carbon that does not decompose is 77 percent (i.e., 0.38/0.49). The remaining 23 percent degrades.

The degradable portion of the carbon is assumed to decay according to first order kinetics. Grass and food scraps are assumed to have a half-life of 5 years; leaves and branches are assumed to have a half-life of 20 years.

For each of the four materials (grass, leaves, branches, food scraps), the stock of carbon in landfills for any given year is calculated according to the following formula:

$$LFC_{i,t} = \sum_{n} W_{i,n} * (1 - MC_i) * ICC_i * \{[CSF_i / ICC_i] + [(1 - (CSF_i / ICC_i)) * e^{-k*(t-n)}]\}$$

where,

t = the year for which carbon stocks are being estimated,
$LFC_{i,t}$ = the stock of carbon in landfills in year *t*, for waste *i* (grass, leaves, branches, food scraps)

$W_{i,n}$ = the mass of waste *i* disposed in landfills in year *n*, in units of wet weight
n = the year in which the waste was disposed, where $1960 \leq n \leq t$
MC_i = moisture content of waste *i*,
ICC_i = the initial carbon content of waste *i*,
CSF_i = the carbon storage factor of waste *i*,
e = the natural logarithm, and
k = the first order rate constant for waste *i*, and is equal to 0.693 divided by the half-life for decomposition.

For a given year *t*, the total stock of carbon in landfills ($TLFC_t$) is the sum of stocks across all four materials. The annual flux of carbon in landfills (F_t) for year *t* is calculated as the change in stock compared to the preceding year:

$$F_t = TLFC_t - TLFC_{t-1}$$

Thus, the carbon placed in a landfill in year n is tracked for each year t through the end of the inventory period (2003). For example, disposal of food scraps in 1960 resulted in depositing about 1,140,000 metric tons of carbon. Of this amount, 16 percent (180,000 metric tons) is persistent; the remaining 84 percent (960,000 metric tons) is degradable. By 1965, half of the degradable portion (480,000 metric tons) decomposes, leaving a total of 660,000 tonnes (the persistent portion, plus the remaining half of the degradable portion).

Continuing the example, by 2003, the total food scraps carbon originally disposed in 1960 had declined to 181,000 metric tons (i.e., virtually all of the degradable carbon had decomposed). By summing the carbon remaining from 1960 with the carbon remaining from food scraps disposed in subsequent years (1961 through 2003), the total landfill carbon from food scraps in 2003 was 29.3 million metric tons. This value is then added to the carbon stock from grass, leaves, and branches to calculate the total landfill carbon stock in 2003, yielding a value of 241.6 million metric tons (as shown in Table 7-19). In exactly the same way total net flux is calculated for forest carbon and harvested wood products, the total net flux of landfill carbon for yard trimmings and food scraps for a given year (Table 7-17) is the difference in the landfill carbon stock for a given year and the stock in the preceding year. For example, the net change in 2003 shown in Table 7-17 (2.7 Tg C) is equal to the stock in 2003 (241.6 Tg C) minus the stock in 2002 (238.9 Tg C).

When applying the carbon storage factor data reported by Barlaz (1998), an adjustment was made to the reported value for leaves, because the carbon storage factor was higher than the initial carbon content. This anomalous result, probably due to errors in the laboratory measurements, was addressed by applying a mass balance calculation, and assuming that (a) the initial carbon content was correctly measured, and (b) the carbon storage factor was incorrect. The same experiment measured not only the persistence of carbon (i.e., the carbon storage factor), but also the yield of methane for each of the individual waste materials (Eleazer et al. 1997). The anaerobic decomposition process results in release of equal molar volumes of CH_4 and CO_2. Thus, to derive a more realistic estimate of the carbon storage factor for leaves, the carbon released in the form of methane during decomposition was multiplied by two (to include the loss of carbon through CO_2, which is generated in approximately equal molar amounts as CH_4), and then subtracted from the initial carbon content of the leaves. This estimate of carbon remaining was used to derive the carbon storage factor (0.46).

Table 7-18: Moisture Content (%), Carbon Storage Factor, Initial Carbon Content (%), Proportion of Initial Carbon Sequestered (%), and Half-Life (years) for Landfilled Yard Trimmings and Food Scraps in Landfills

	Yard Trimmings			Food Scraps
Variable	**Grass**	**Leaves**	**Branches**	
Moisture Content (% H_2O)	70	30	10	70
CSF (kg C sequestered / dry kg waste)	0.32	0.46[a]	0.38	0.08
Initial Carbon Content (%)	45	49	49	51
Proportion of initial carbon sequestered (%)	71	94	77	16
Half-life (years)	5	20	20	5

[a] Adjusted using CH_4 yields in Eleazer et al. (1997).

Table 7-19: Carbon Stocks in Yard Trimmings and Food Scraps in Landfills (Tg C)

Carbon Pool	1990	1996	1997	1998	1999	2000	2001	2002	2003
Yard Trimmings	**167.8**	**196.5**	**199.3**	**201.9**	**204.2**	**206.2**	**208.2**	**210.2**	**212.3**
Grass	18.8	21.7	21.9	22.2	22.4	22.5	22.7	22.9	23.1
Leaves	78.7	92.9	94.3	95.7	97.0	98.0	99.1	100.2	101.3
Branches	70.3	81.9	83.0	84.0	84.9	85.6	86.4	87.1	87.9
Food Scraps	**20.3**	**24.0**	**24.7**	**25.5**	**26.3**	**27.1**	**27.9**	**28.6**	**29.3**
Total Carbon Stocks	**188.1**	**220.5**	**224.0**	**227.4**	**230.5**	**233.3**	**236.1**	**238.9**	**241.6**

Note: Totals may not sum due to independent rounding.

Uncertainty

The estimation of carbon storage in landfills is directly related to the following yard trimming and food scrap data and factors: disposal in landfills per year (tons of carbon), initial carbon content, moisture content, decomposition rate (half-life), and proportion of carbon stored. The carbon storage landfill estimates are also a function of the composition of the yard trimmings (i.e. the proportions of grass, leaves and branches in the yard trimmings mixture). There are uncertainties associated with each of these factors.

The uncertainty ranges were assigned based on expert judgment and are assumed to be uniformly distributed around the inventory estimate (e.g., $\pm$10 percent), except for the values for decomposition rate, proportion of carbon stored, and moisture content for branches.

The uncertainty ranges associated with the input variables for the proportion of grass and leaves in yard trimmings, as well as the initial carbon content and moisture content for grass, leaves, and food scraps (all expressed as percentages in the calculations for the inventory) were plus or minus 10 percent. For the moisture content of branches (where the inventory estimate is 10 percent), the uncertainty range was assumed to be 5 to 30 percent.

The uncertainty ranges associated with the disposal of grass, leaves, branches, and food scraps were bound at 50 percent to 150 percent times the inventory estimates. The half-life of grass and food scraps were assumed to range from 1 to 20 years, and the half-lives of leaves and branches were assumed to range from 5 to 30 years. Finally, the proportion of carbon stored in grass, leaves, branches, and food scraps was assumed to vary by the addition of 20 percent or subtraction of 10 percent from the best estimate, with an upper bound of 100 percent and a lower bound of 0 percent.

A Monte Carlo (Tier 2) uncertainty analysis was then applied to estimate the overall uncertainty of the sequestration estimate. The results of the Tier 2 quantitative uncertainty analysis are summarized in Table 7-20. Total yard trimmings and food scraps CO_2 flux in 2003 was estimated to be between -17.5 and -7.0 Tg CO_2 Eq. at a 95 percent confidence level (or 19 of 20 Monte Carlo Stochastic Simulations). This indicates a range of 73 percent below to 31 percent above the 2003 flux estimate of -10.1 Tg CO_2 Eq.

Table 7-20: Tier 2 Quantitative Uncertainty Estimates for CO_2 Flux from Yard Trimmings and Food Scraps in Landfills (Tg CO_2 Eq. and Percent)

Source	Gas	2003 Flux Estimate (Tg CO_2 Eq.)	Uncertainty Range Relative to Flux Estimate[a] (Tg CO_2 Eq.)		(%)	
			Lower Bound	Upper Bound	Lower Bound	Upper Bound
Yard Trimmings	CO_2	(7.5)	(11.3)	(4.4)	-51%	+41%
Food Scraps	CO_2	(2.6)	(8.9)	(1.2)	-246%	+54%
Total	**CO_2**	**(10.1)**	**(17.5)**	**(7.0)**	**-73%**	**+31%**

[a]Range of flux estimates predicted by Monte Carlo Stochastic Simulation for a 95 percent confidence interval. Negative values indicate net carbon storage (positive values denote emissions).

The uncertainty of the landfilled carbon storage estimate arises from the disposal data and the factors applied to the following data.

Disposal per Year (tons of carbon)

A source of uncertainty affecting CO_2 sequestration is the estimate of the tonnage of yard trimmings and food scraps which are disposed of in landfills each year. Of all the individual inputs tested for sensitivity in the uncertainty analysis, net carbon storage in landfills is most sensitive to the estimate of the food scrap disposal rate. The estimates for yard trimming and food scrap disposal in landfills are determined using data from the EPA (1999, 2002, 2003) estimates of materials generated, discarded, and combusted, which carry considerable uncertainty associated with the wastestream sampling methodology used to generate them.

Moisture Content and Initial Carbon Content

Moisture content, and to a lesser extent carbon content, vary widely. Moisture content for a given sample of waste can be affected by the precipitation conditions when the waste is placed at the curb for collection, as well as the status and condition of the landfill cover. Carbon content (on a dry weight basis) is a function of the specific waste constituents (e.g., oak leaves versus pine needles or banana peels versus bacon grease), which in turn vary temporally, geographically, and demographically (i.e., characteristics of households in the wasteshed).

Decomposition Rate

Although several investigators have made estimates of the decomposition rate of mixed solid waste in a landfill environment, there are no known studies of decomposition rates for individual materials in actual landfills, and thus the inventory estimate is based on assumed values. The uncertainty analysis indicates that the results are sensitive to decomposition rates, especially the food scraps half-life, and thus the decomposition rates introduce considerable uncertainty into the analysis.

Proportion of Carbon Stored

The estimate of the proportion of carbon stored is based on a set of experiments measuring the amount of carbon persisting in conditions promoting decomposition. Because these experiments have only used conditions conducive to decomposition, they are more likely to underestimate than to overestimate carbon storage. Thus, the uncertainty analysis used asymmetrical values (up to 10 percent less storage, up to 20 percent more storage) as inputs.

Several of the planned improvements to the analysis, described later in this section, are intended to reduce the uncertainty associated with these factors.

Recalculations Discussion

While conducting quality control procedures, it was found that the 1994 value for the amount of discards that are landfilled (or disposed of by means other than combustion) had been incorrectly recorded. The error was corrected, resulting in reduced carbon storage values by less than 1.0 Tg C for the years 1994 through 2003. Overall, this change, in combination with historical data revisions, resulted in an average annual decrease of 0.2 Tg CO_2 Eq. (0.7 percent) in carbon sequestration from yard trimming and food scraps over the period 1990 through 2002.

Planned Improvements

As noted above, the estimates presented in this section are driven by a small carbon storage factor data set, and some of these measurements (especially for leaves) deserve close scrutiny. There are ongoing efforts to conduct a re-analysis of the leaves experiment, using the same techniques as in the original experiments cited, and future work may evaluate the potential contribution of inorganic carbon to landfill sequestration and to assure consistency between the estimates of carbon storage described in this chapter and the estimates of landfill CH_4 emissions described in the Waste chapter.

Changes in Carbon Stocks in Urban Trees (IPCC Source Category 5E1)

Urban forests constitute a significant portion of the total U.S. tree canopy cover (Dwyer et al. 2000). Urban areas (cities, towns, and villages), which cover 3.5 percent of the continental United States, are estimated to contain about 3.8 billion trees. With an average tree canopy cover of 27.1 percent, urban areas account for approximately 3 percent of total tree cover in the continental United States (Nowak et al. 2001). Trees in urban areas of the continental United States were estimated by Nowak and Crane (2002) to account for an average annual net sequestration of 58.7 Tg CO_2 Eq. (16 Tg C). These data were collected throughout the 1990s, and have been applied to the entire time series in this report (see Table 7-21). Annual estimates of CO_2 flux have not been developed, but are believed to be relatively constant from 1990 through 2003.

Net carbon flux from urban trees is proportionately greater on an area basis than that of forests. This trend is primarily the result of different net growth rates in urban areas versus forests—urban trees often grow faster than forest trees because of the relatively open structure of the urban forest (Nowak and Crane 2002). Also, areas in each case are accounted for differently. Because urban areas contain less tree coverage than forest areas, the carbon storage per hectare of land is in fact smaller for urban areas. However, urban tree reporting occurs on a per unit tree cover basis (tree canopy area), rather than total land area. Urban trees therefore appear to have a greater carbon density than forested areas (Nowak and Crane 2002).

Table 7-21: Net C Flux from Urban Trees (Tg CO_2 Eq. and Tg C)

Year	Tg CO_2 Eq.	Tg C
1990	(58.7)	(16)
1997	(58.7)	(16)
1998	(58.7)	(16)
1999	(58.7)	(16)
2000	(58.7)	(16)
2001	(58.7)	(16)
2002	(58.7)	(16)
2003	(58.7)	(16)

Note: Parentheses indicate net sequestration.

Methodology

The methodology used by Nowak and Crane (2002) is based on average annual estimates of urban tree growth and decomposition, which were derived from field measurements and data from the scientific literature, urban area estimates from U.S. Census data, and urban tree cover estimates from remote sensing data. This approach is consistent with the default IPCC methodology in the IPCC *LULUCF Good Practice Guidance* (IPCC 2003), although sufficient data are not yet available to determine interannual changes in carbon stocks in the living biomass of urban trees.

Nowak and Crane (2002) developed estimates of annual gross carbon sequestration from tree growth and annual gross carbon emissions from decomposition for ten U.S. cities: Atlanta, GA; Baltimore, MD; Boston, MA; Chicago, IL; Jersey City, NJ; New York, NY; Oakland, CA; Philadelphia, PA; Sacramento, CA; and Syracuse, NY. The gross carbon sequestration estimates were derived from field data that were collected in these ten cities during the period from 1989 through 1999, including tree measurements of stem diameter, tree height, crown height, and crown width, and information on location, species, and canopy condition. The field data were converted to annual gross carbon sequestration rates for each species (or genus), diameter class, and land-use condition (forested, park-like, and open growth) by applying allometric equations, a root-to-shoot ratio, moisture contents, a carbon content of 50 percent (dry weight basis), an adjustment factor to account for smaller aboveground biomass volumes (given a particular diameter) in urban conditions compared to forests, an adjustment factor to account for tree condition (fair to excellent, poor, critical, dying, or dead), and annual diameter and height growth rates. The annual gross carbon sequestration rates for each species (or genus), diameter class, and land-use condition were then scaled up to city estimates using tree population information. The field data from the 10 cities, some of which are unpublished, are

described in Nowak and Crane (2002) and references cited therein. The allometric equations were taken from the scientific literature (see Nowak 1994, Nowak et al. 2002), and the adjustments to account for smaller volumes in urban conditions were based on information in Nowak (1994). A root-to-shoot ratio of 0.26 was taken from Cairns et al. (1997), and species- or genus-specific moisture contents were taken from various literature sources (see Nowak 1994). Adjustment factors to account for tree condition were based on percent crown dieback (Nowak and Crane 2002). Tree growth rates were also taken from existing literature. Average diameter growth was based on the following sources: estimates for trees in forest stands came from Smith and Shifley (1984); estimates for trees on land uses with a park-like structure came from deVries (1987); and estimates for more open-grown trees came from Nowak (1994). Formulas from Fleming (1988) formed the basis for average height growth calculations.

Annual gross carbon emission estimates were derived by applying estimates of annual mortality and condition, and assumptions about whether dead trees were removed from the site, to carbon stock estimates. These values were derived as intermediate steps in the sequestration calculations, and different decomposition rates were applied to dead trees left standing compared with those removed from the site. The annual gross carbon emission rates for each species (or genus), diameter class, and condition class were then scaled up to city estimates using tree population information. Estimates of annual mortality rates by diameter class and condition class were derived from a study of street-tree mortality (Nowak 1986). Assumptions about whether dead trees would be removed from the site were based on expert judgment of the authors. Decomposition rates were based on literature estimates (Nowak and Crane 2002).

Annual net carbon sequestration estimates were derived for seven of the ten cities by subtracting the annual gross emission estimates from the annual gross sequestration estimates.[9] National annual net carbon sequestration by urban trees was estimated from the city estimates of gross and net sequestration, and urban area and urban tree cover data for the contiguous United States. The urban areas are based on 1990 U.S. Census data, which define "urban land" as areas having a population density greater than 1,000 people per square mile and adjacent urban places, with predefined political boundaries, having a population total greater than 2,500. Therefore, urban encompasses most cities, towns, and villages (i.e., it includes both urban and suburban areas). The gross and net carbon sequestration values for each city were divided by each city's area of tree cover to determine the average annual sequestration rates per unit of tree area for each city. The median value for gross sequestration (0.30 kg C/m^2-year) was then multiplied by an estimate of national urban tree cover area (76,151 km^2) to estimate national annual gross sequestration. To estimate national annual net sequestration, the estimate of national annual gross sequestration was multiplied by the average of the ratios of net to gross sequestration for those cities that had both estimates. The average of these ratios is 0.70. The urban tree cover area estimates for each of the 10 cities and the contiguous United States were obtained from Dwyer et al. (2000) and Nowak et al. (2001).

Table 7-22: Carbon Stocks (Metric Tons C), Annual Carbon Sequestration (Metric Tons C/yr), Tree Cover (Percent), and Annual Carbon Sequestration per Area of Tree Cover (kg C/m^2 cover-yr) for Ten U.S. Cities

City	Carbon Stocks	Gross Annual Sequestration	Net Annual Sequestration	Tree Cover	Gross Annual Sequestration per Area of Tree Cover	Net Annual Sequestration per Area of Tree Cover
New York, NY	1,225,200	38,400	20,800	20.9	0.23	0.12
Atlanta, GA	1,220,200	42,100	32,200	36.7	0.34	0.26
Sacramento, CA	1,107,300	20,200	NA	13.0	0.66	NA
Chicago, IL	854,800	40,100	NA	11.0	0.61	NA
Baltimore, MD	528,700	14,800	10,800	25.2	0.28	0.20
Philadelphia, PA	481,000	14,600	10,700	15.7	0.27	0.20
Boston, MA	289,800	9,500	6,900	22.3	0.30	0.22
Syracuse, NY	148,300	4,700	3,500	24.4	0.30	0.22
Oakland, CA	145,800	NA	NA	21.0	NA	NA

[9] Three cities did not have net estimates.

Jersey City, NJ	19,300	800	600	11.5	0.18	0.13

NA = not analyzed

Uncertainty

The only quantifiable uncertainty associated with changes in C stocks in urban trees was sampling, as reported by Nowak and Crane (2002). The average standard deviation for urban tree carbon storage was 27 percent of the mean carbon storage on an area basis. Additionally, a 5 percent uncertainty was associated with national urban tree covered area. These estimates are based on field data collected in ten U.S. cities, and uncertainty in these estimates increases as they are scaled up to the national level.

There is additional uncertainty associated with the biomass equations, conversion factors, and decomposition assumptions used to calculate carbon sequestration and emission estimates (Nowak et al. 2002). These results also exclude changes in soil carbon stocks, and there may be some overlap between the urban tree carbon estimates and the forest tree carbon estimates. However, both the omission of urban soil carbon flux and the potential overlap with forest carbon are believed to be relatively minor (Nowak 2002). Because these are inestimable, they are not quantified as part of this analysis.

The results of the Tier 1 quantitative uncertainty analysis are summarized in Table 7-23. Net C flux from changes in C stocks in urban trees was estimated to be between -80.3 and -37.0 Tg CO_2 Eq. at a 95 percent confidence level. This indicates a range of 37 percent above and below the 2003 flux estimate of -58.7 Tg CO_2 Eq.

Table 7-23: Tier 1 Quantitative Uncertainty Estimates for Net C Flux from Changes in Carbon Stocks in Urban Trees (Tg CO_2 Eq. and Percent)

Source	Gas	2003 Flux Estimate (Tg CO_2 Eq.)	Uncertainty (%)	Uncertainty Range Relative to 2003 Flux Estimate (Tg CO_2 Eq.)	
				Lower Bound	Upper Bound
Changes in C Stocks in Urban Trees	CO_2	(58.7)	37%	(80.3)	(37.0)

Note: Parentheses indicate net sequestration.

QA/QC and Verification

The net carbon flux resulting from urban trees was calculated using estimates of gross and net carbon sequestration estimates for urban trees and urban tree coverage area found in literature. The validity of these data for their use in this section of the Inventory was evaluated through correspondence established with an author of the papers. Through the correspondence, the methods used to collect the urban tree sequestration and area data were further clarified and the use of these data in the Inventory was reviewed and validated (Nowak 2002).

Planned Improvements

Some sources indicate a reduction in urban tree coverage in the United States over the Inventory period of approximately 21 percent over the last 10 years (AF 2004). However, because the methods for making this assertion have not yet been made available and their definition of urban land is unclear, the veracity and potential application of this estimate cannot currently be established. Because the magnitude of the urban tree greenhouse gas sink in the United States is not insignificant, identifying changes in this sector is considered a priority and is being actively pursued for inclusion in future Inventories. Should this diminishment prove to be accurate, it could mean the urban tree sink estimates will need to be significantly revised.

Changes in Soil Carbon Stocks (IPCC Source Category 5E1)

Given the lack of available land use information relevant to this particular IPCC source category, it is not possible to quantify the CO_2 flux from soils in Settlements Remaining Settlements at this time.

N_2O Fluxes from Soils (IPCC Source Category 5E1)

Of the fertilizers applied to soils in the United States, approximately 10 percent are applied to lawns, golf courses, and other landscaping occurring within settled areas. Application rates are less than those occurring on cropped soils, and therefore account for a smaller proportion of total U.S. soil N_2O emissions per unit area. In 2003, N_2O emissions from this source were 6.0 Tg CO_2 Eq. (19.4 Gg N_2O). There was an overall increase of 9 percent over the thirteen year period due to a general increase in the application of synthetic fertilizers. Interannual variability in these emissions is directly attributable to interannual variability in total synthetic fertilizer consumption in the United States.

Emissions from this source are summarized in Table 7-24.

Table 7-24: N_2O Fluxes from Soils in Settlements Remaining Settlements (Tg CO_2 Eq.)

Settlements Remaining Settlements: N_2O Fluxes from Soils	**1990**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**
Tg CO_2 Eq.	5.5	6.1	6.1	6.2	6.0	5.8	6.0	6.0
Gg	17.9	19.8	19.8	19.9	19.3	18.7	19.4	19.4

Methodology

Estimates of direct N_2O emissions from soils in settlements were based on the amount of N applied to turf grass annually through the application of synthetic commercial fertilizers. Nitrogen applications to turf grass are assumed to be 10 percent of the total synthetic fertilizer used in the United States (Qian 2004). Total synthetic fertilizer applications were derived from fertilizer statistics (TVA 1991, 1992, 1993, 1994; AAPFCO 1995, 1996, 1997, 1998, 1999, 2000b, 2002, 2003, 2004) and a recent AAPFCO database (AAPFCO 2000a). Unvolatilized N applied to turf grass was multiplied by the IPCC default emission factor (1.25 percent) to estimate direct N_2O emissions. The volatilized and leached/runoff proportion, calculated with the IPCC default volatilization factor of 10 percent and 30 percent, respectively, for synthetic fertilizers, was included with the total N contributions to indirect emissions, as reported in the N_2O Emissions from Agricultural Soil Management source category of the Agriculture sector.

Uncertainty

The amount of N_2O emitted from settlements depends not only on N inputs, but also on a large number of variables, including organic carbon availability, O_2 partial pressure, soil moisture content, pH, temperature, and irrigation/watering practices. The effect of the combined interaction of these variables on N_2O flux is complex and highly uncertain. The IPCC default methodology used here does not incorporate any of these variables and only accounts for variations in national fertilizer application rates. All settlement soils are treated equivalently under this methodology. Furthermore, only synthetic fertilizers are captured, so applications of organic fertilizers are not accounted for here. Uncertainties exist in both the fertilizer application rates and the emission factors used to derive emission estimates.

The 95 percent confidence interval for the IPCC's default emission factor for synthetic fertilizer applied to soil ranges from 0.25 to 6 percent, according to Chapter 4 of IPCC (2000). While a Tier 1 analysis should be generated from a symmetrical distribution of uncertainty around the emission factor, an asymmetrical distribution was imposed here to account for the fact that the emission used was not the mean of the range given by IPCC. Therefore, an upper bound of 480 percent and a lower bound of 80 percent were assigned to the emission factor. The higher uncertainty percentage is shown below, but the lower bound reflects a truncated distribution. The uncertainty in the amount of synthetic fertilizer N applied to settlement soils was conservatively estimated to be 50 percent (Qian 2004). The results of the Tier 1 quantitative uncertainty analysis are summarized in Table 7-25. N_2O emissions from soils in settlements remaining settlements in 2003 were estimated to be between 0.3 and 35.1 Tg CO_2 Eq. at a 95 percent confidence level. This indicates a range of 94 percent below to 483 percent above the 2003 emission estimate of 6.0 Tg CO_2 Eq.

Table 7-25: Tier 1 Quantitative Uncertainty Estimates of N_2O Emissions from Soils in Settlements Remaining Settlements (Tg CO_2 Eq. and Percent)

IPCC Source Category	Gas	Year 2003 Emissions (Tg CO_2 Eq.)	Uncertainty (%)	Uncertainty Range Relative to 2003 Emission Estimate (Tg CO_2 Eq.)	
				Lower Bound	Upper Bound
Settlements Remaining Settlements: N_2O Fluxes from Soils	N_2O	6.0	94 to 483%	0.3	35.1

Recalculations Discussion

The current Inventory presents N_2O emissions from soils in settlements separately for the first time. Previously, N_2O emissions from this source were included with N_2O emissions from agricultural soils. The net effect of separating this source from agricultural soils is to reduce emissions reported from that source. However, because the methods for reporting that source category have changed significantly this year, it is not possible to isolate the magnitude of change caused by this recalculation alone on the overall differences in N_2O emissions from agricultural soils.

Planned Improvements

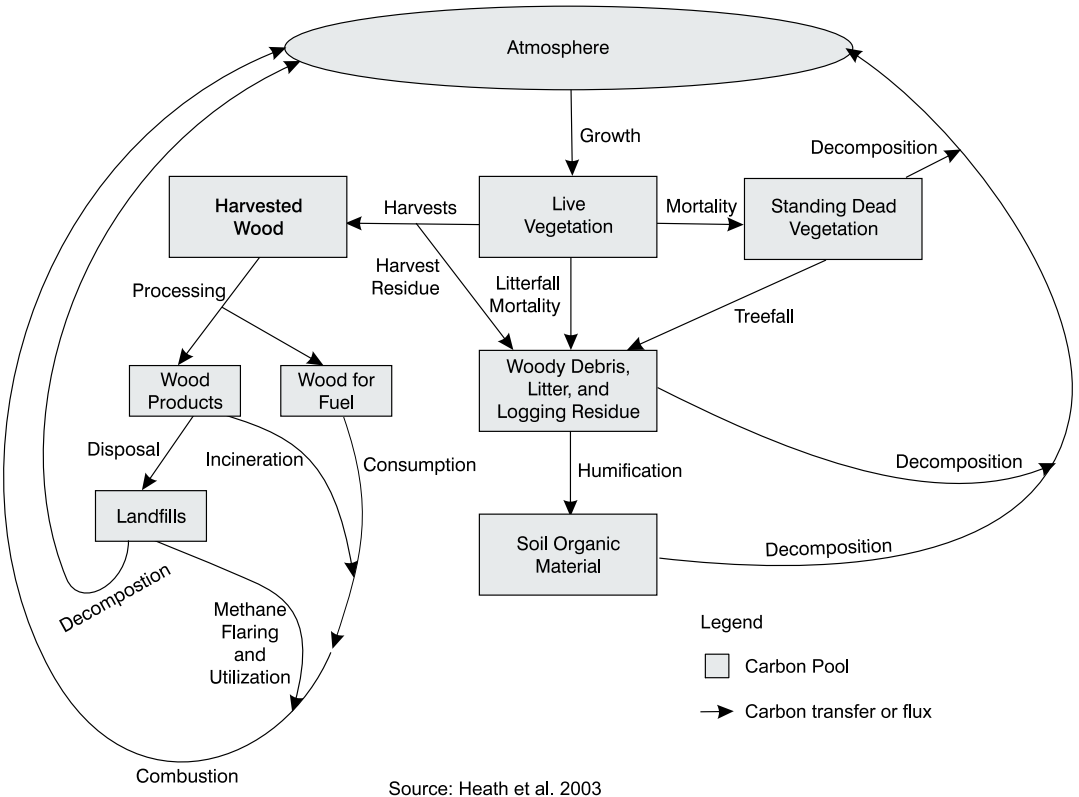
The process-based model DAYCENT, which was used to estimate N_2O emissions from cropped soils this year, could also be used to simulate emissions as well as volatilization and leaching/runoff from settlements. DAYCENT has been parameterized to simulate turf grass. State-level settlement area data is available from the National Resource Inventory. Future inventories will include DAYCENT simulations to estimate emissions from settlements.

7.6. Lands Converted to Settlements (Source Category 5E2)

Land-use change is constantly occurring, and land under a number of uses undergoes urbanization in the United States each year. However, data on the amount of land converted to settlements is currently lacking. Given the lack of available information relevant to this particular IPCC source category, it is not possible to quantify CO_2 or N_2O fluxes from lands converted to settlements at this time.

Figure 7-1

Forest Sector Carbon Pools and Flows



Source: Heath et al. 2003

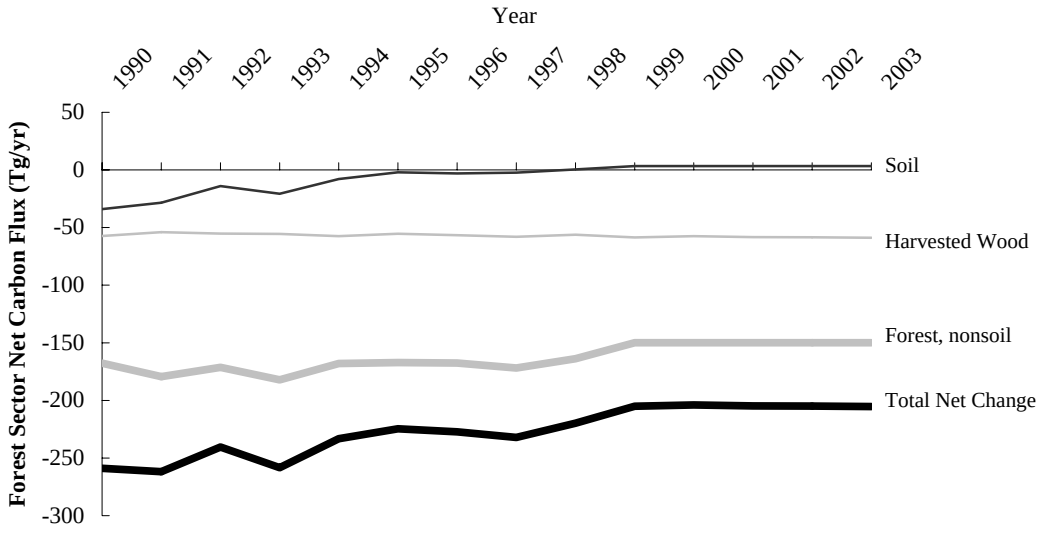
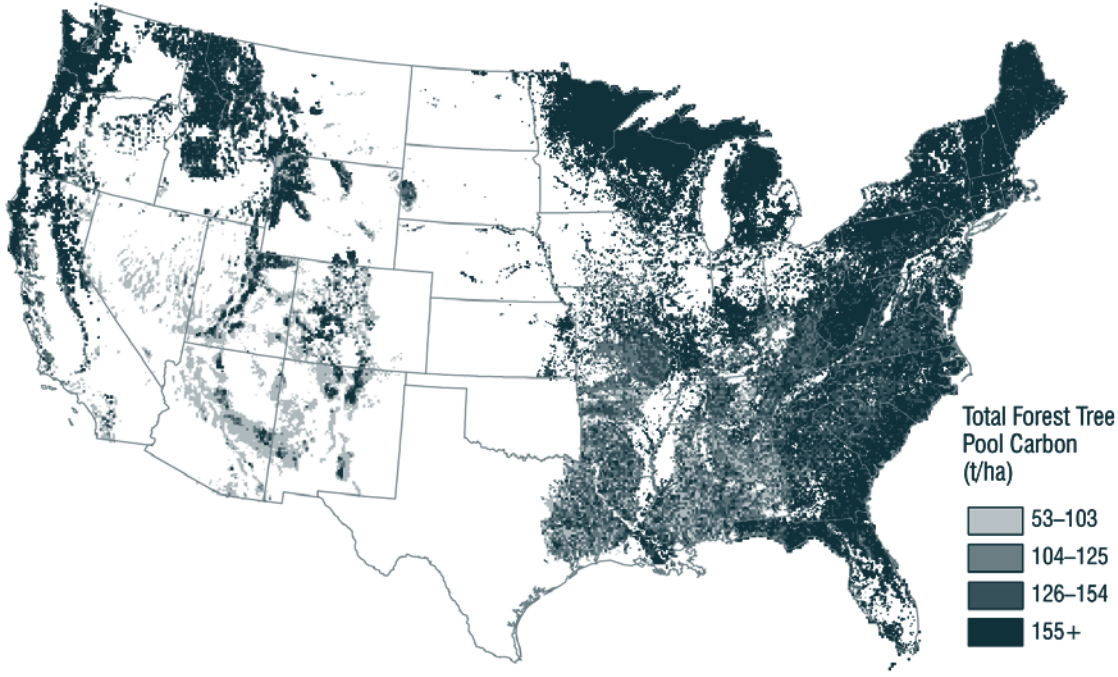


Figure 7-2: Estimates of Net Annual Changes in Carbon Stocks for Major Carbon Pools

Figure 7-3

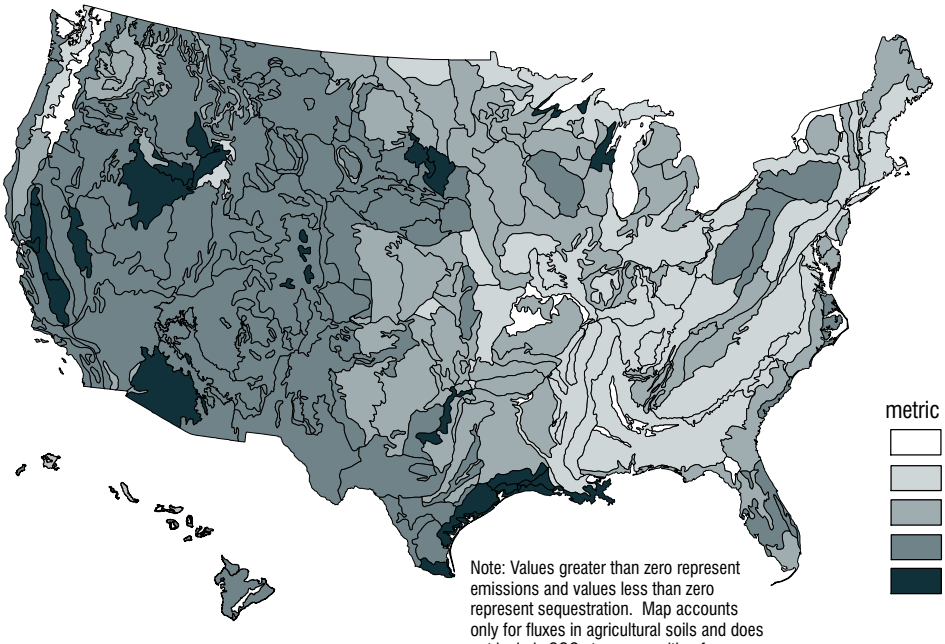
Average Carbon Density in the Forest Tree Pool in the Conterminous U.S. During 2004



Note: Estimates are based on forest inventory data as described in the text.

Figure 7-4

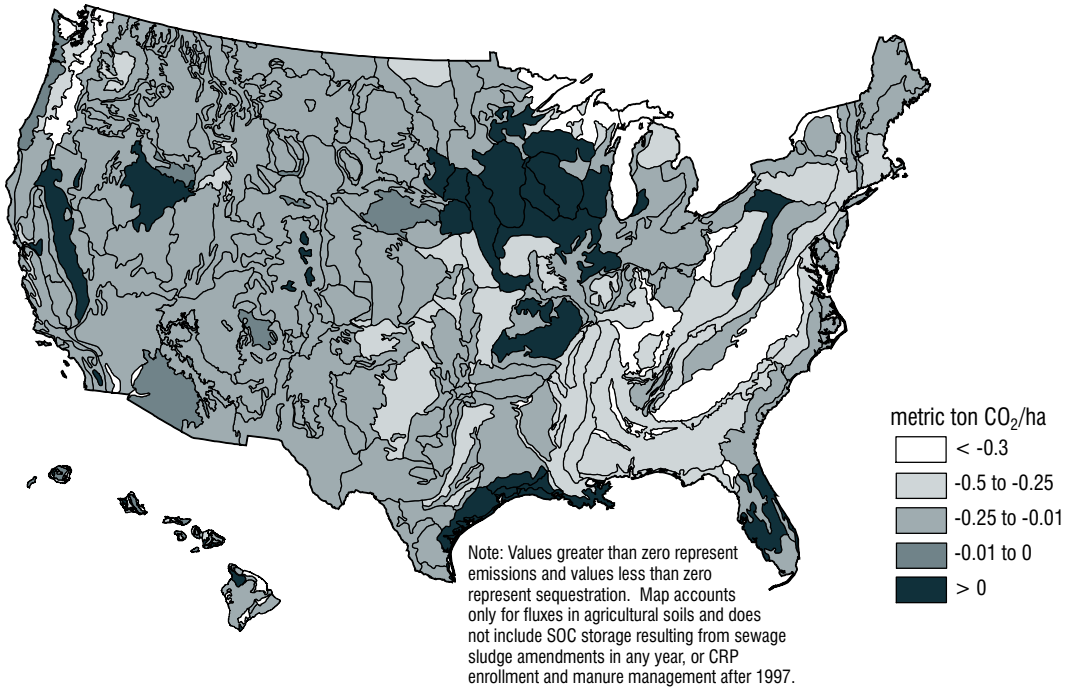
Net Annual CO_2 Flux, per Hectare, From Mineral Soils Under Agricultural Management, 1990–1992



This map shows the spatial variability in net annual carbon dioxide flux from mineral soils for the year 1990 through 1992. The color assigned to each polygon represents the average annual flux per hectare for the area of managed mineral soils in that polygon.

Figure 7-5

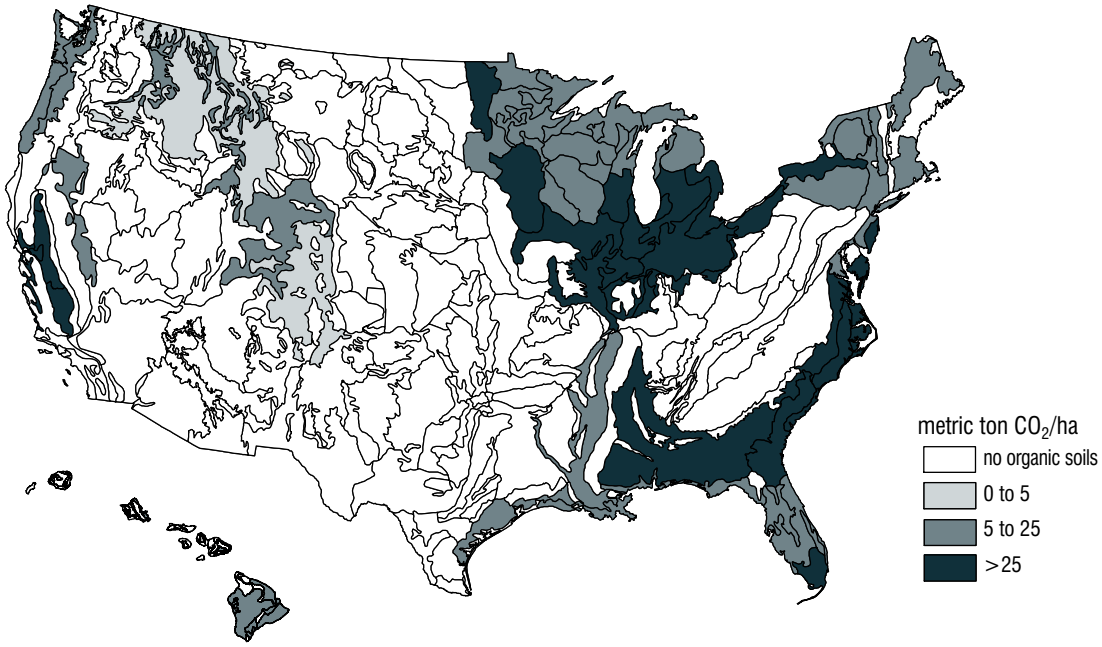
Net Annual CO_2 Flux, per Hectare, From Mineral Soils Under Agricultural Management, 1993–2003



This map shows the spatial variability in net annual carbon dioxide flux from mineral soils for the year 1993 through 2003. The color assigned to each polygon represents the average annual flux per hectare for the area of managed mineral soils in that polygon.

Figure 7-6

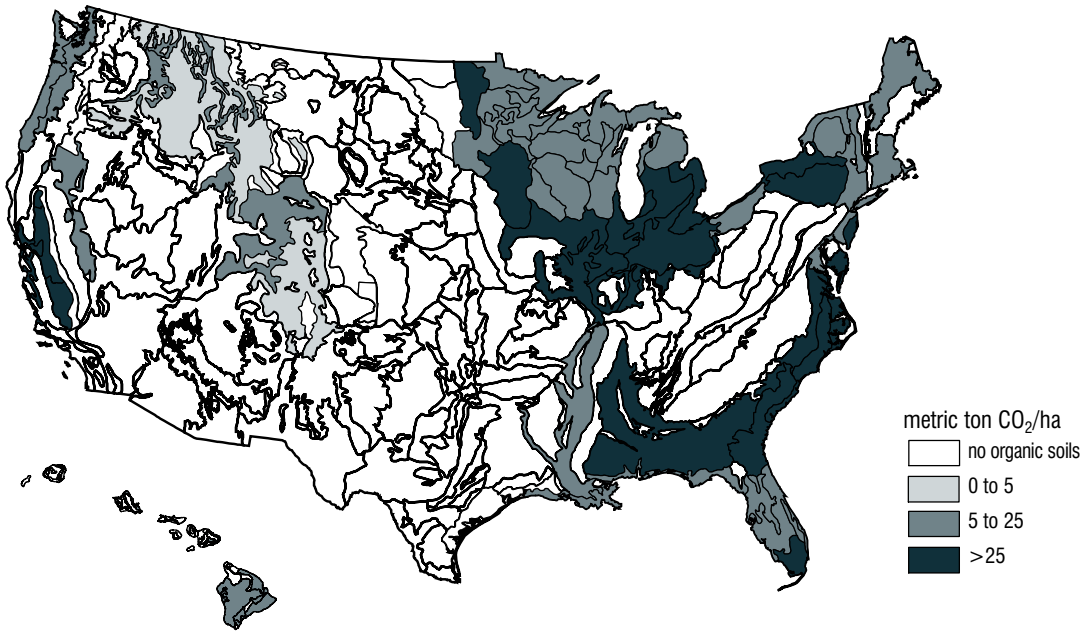
Net Annual CO_2 Flux, per Hectare, From Organic Soils Under Agricultural Management, 1990–1992



This map shows the spatial variability in net annual carbon dioxide flux from organic soils for the year 1990 through 1992.
The color assigned to each polygon represents the average annual flux per hectare for the area of managed organic soils in that polygon.

Figure 7-7

Net Annual CO_2 Flux, per Hectare, From Organic Soils Under Agricultural Management, 1993–2003



This map shows the spatial variability in net annual carbon dioxide flux from organic soils for the year 1993 through 2003.
The color assigned to each polygon represents the average annual flux per hectare for the area of managed organic soils in that polygon.